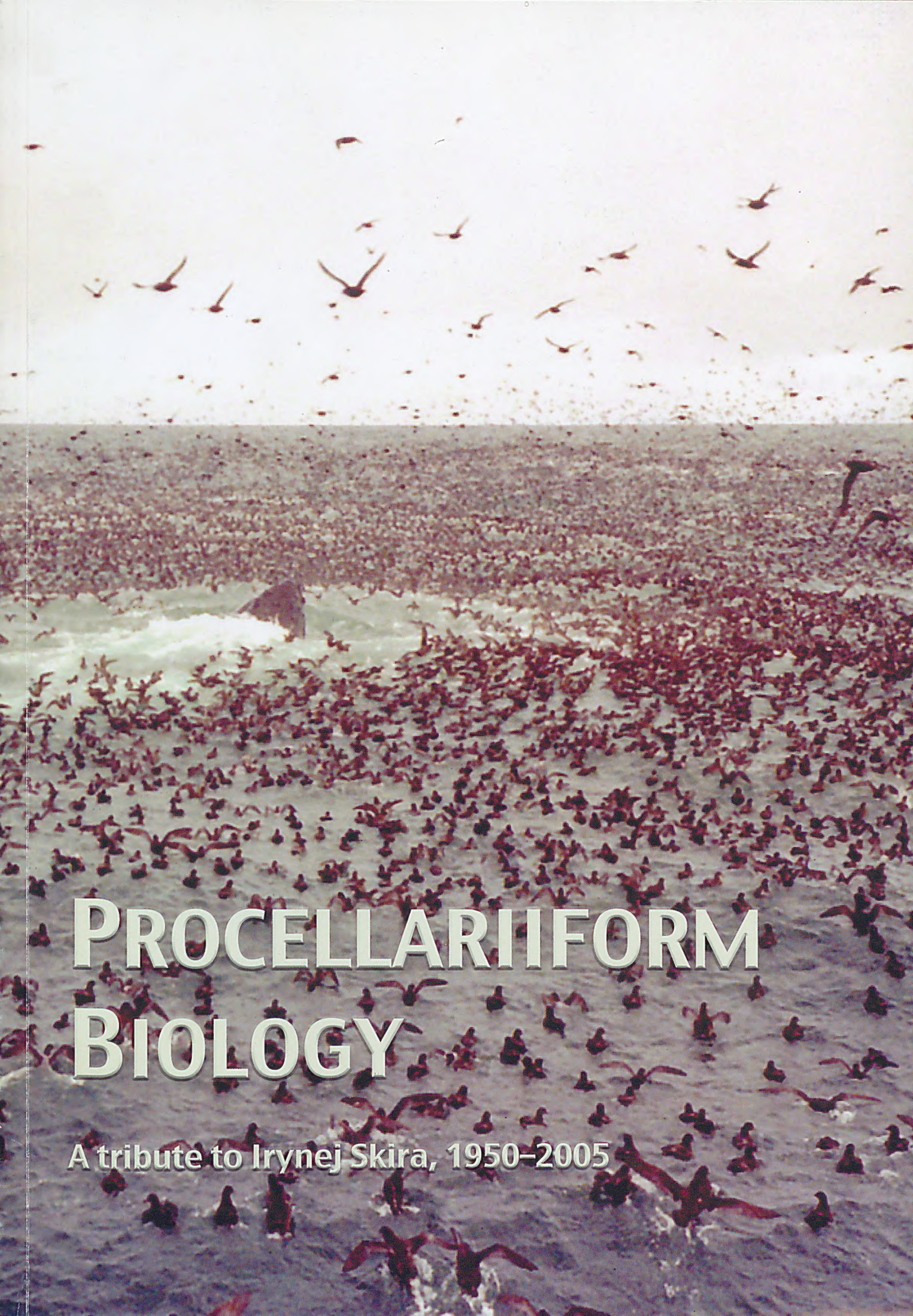

# PROCELLARIIFORM BIOLOGY

A tribute to Irynej Skira, 1950–2005

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THE

ROYAL

SOCIETY

OF TASMANIA

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Henrik Moller and Stuart Bradley

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Cover photograph — Part of an aggregation of Short-tailed Shearwaters, *Puffinus tenuirostris*, and Humpback Whales, *Megaptera novaeangliae* in the Bering Sea. Photograph graciously provided by Mike Brittain of Seward, Alaska, and was taken when he was working with the North Gulf Oceanic Society on Killer, *Orcinus orca*, and Humpback whales.

*Statements of opinions expressed in papers in this volume reflect the views of the author(s) and are not necessarily those of the Royal Society of Tasmania.*

PAPERS AND PROCEEDINGS OF THE ROYAL SOCIETY OF TASMANIA
VOLUME 142, PART 1

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# AUSTRAL SEABIRDS: CHALLENGES AND OPPORTUNITIES FOR RESEARCH AND CONSERVATION

by Ben Knight, Henrik Moller, Stuart Bradley and Margaret Davies

**Knight, B., Moller, H., Bradley, S. & Davies, M.** 2008 (31:xi): Austral seabirds: challenges and opportunities for research and conservation. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 1–8. ISSN 0080-4703. Centre for the Study of Agriculture, Food and Environment, University of Otago, PO Box 56, Dunedin 9054, New Zealand (BK, HM*), Department of Biological Sciences and Biotechnology, Murdoch University, South Street, Murdoch, Western Australia 6150, Australia (SB), Royal Society of Tasmania, GPO Box 1166, Hobart, Tasmania 7001 Australia (MD) * Author for correspondence. Email: henrik.moller@otago.ac.nz

This paper draws together the themes of papers on procellariiform biology contained within this special issue of the *Papers and Proceedings of the Royal Society of Tasmania* which is a tribute to Irynej Skira. The role of these birds as generalised biomonitors of marine ecosystem health as well as their important interactions with commercial fisheries and human societies are major considerations. Seabird conservation faces challenges from introduced pest species, loss of habitat to urban development, marine pollution and climate and oceanographic changes. Studies are hampered by the difficulties of dealing with birds in remote areas (that are therefore expensive to study), a paucity of funds and problems of overlapping national and international jurisdictions in their individual home ranges or on their migration routes. There is a real need for long-term studies of seabirds because they have delayed breeding systems, a slow rate of reproduction, long life-spans and high adult survival. Such knowledge is important in understanding their population dynamics, the effects of changing climate, the impacts of commercial fishing, pollution, breeding habitat loss and the harvesting of chicks. The value of the long-term studies of Short-tailed Shearwaters, *Puffinus tenuirostris,* on Fisher Island, Tasmania, is considered in the light of the importance of research investment in basic curiosity-driven research. Neglect of long-term studies and the diminishing role of local scientific journals in the dissemination of baseline data on which to build generalities are the result of changing priorities of government and other funding agencies towards short-term output-based assessment models. Our present inability to answer the basic question of what determines seabird population abundance and distribution highlights a lack of fundamental population data in Austral seabird research. This must be addressed so that a mixture of comparative, experimental and modelling studies can be mobilised to complement descriptive studies.

**Key Words: Procellariiformes, seabird ecology, conservation management, research funding, long-term research.**

## INTRODUCTION

The research included in this special issue of the *Papers and Proceedings of the Royal Society of Tasmania* congregates around a central focus on seabirds that breed in coastal regions and outlying islands of Australia (eight papers) and New Zealand (seven papers). This territory is of paramount importance for seabirds globally, with worldwide seabird species density peaking in the north of New Zealand and south of Australia (Chown *et al.* 1998, Taylor 2000).

Despite the diversity and ecological importance of seabirds in our region, most Austral ornithology has contributed short-term and somewhat fragmentary information from a wide variety of breeding areas. An exception, and one of the longest seabird studies in the Southern Hemisphere, is that of the Short-tailed Shearwater, *Puffinus tenuirostris* (Temminck, 1835), population on Fisher Island, in Bass Strait. The study was initiated by D.L. (Dom) Serventy of the CSIRO in collaboration with the Tasmanian Wildlife Authority in 1947 to respond to concern about the impact of harvesting levels on this species. Fisher Island was selected because of its small size (0.8 ha), and its accessibility (250 m from the local township). This allowed all breeding shearwaters at the colony (two to three hundred) to be marked individually and monitored each year. These logistical benefits undoubtedly played a pivotal role in enabling the study to continue with unchanged protocols over its protracted life span, but came at the price of uncertainty as to how well the Fisher Island populations represented the generalities of the much larger colonies (hundreds of thousands) on other islands. Dom Serventy's reputation as a leading seabird biologist, and his many contacts, established an international web of support for this farsighted project. In the early 1970s Dr Irynej Skira, who had become Serventy's associate in the project, took over the work of banding the shearwaters on Fisher Island and some other of Tasmania's Bass Strait islands. Important contributions followed by Professor Ron Wooller.

The research team was able to gain fundamental knowledge of the species' demography and breeding ecology, including breeding success, site and mate fidelity, and longevity (Ryan 2008). The team's annual population estimates helped determine sustainable harvest of the species and guide indigenous use. Over 33 years while employed by the Tasmanian Government, Dr Skira's commitment and quiet determination resulted in a long-term study on a scale that is rare in the world's seabird research. His work resulted in over 40 scientific papers, technical reports, his doctorate, countless media articles and presentations, and forged links with seabird scientists across the globe (Ryan 2008, Sally Bryant pers. comm.).

This memorial issue pays tribute to Dr Skira's work, and illustrates the current state and continuing scope of seabird research in Australasia. The contributing studies deal with wide-ranging aspects of seabird ecology, including population dynamics (Newman *et al.* 2008b, Scott *et al.* 2008), breeding and colonial behaviour (Bancroft 2008, Powell *et al.* 2008), island biodiversity (Copson & Brothers 2008, Brothers & Ledingham 2008, Bancroft *et al.* 2008), foraging behaviour (Taylor 2008, MacLeod *et al.* 2008), human impacts (Rosier & Dyer 2008, Hutton *et al.* 2008, Waugh *et al.* 2008), migration (Oka 2008), pest management (Brothers & Bone 2008), indigenous use and environmentality (Kitson & Moller 2008, Lyver *et al.* 2008), and climate change (Brothers & Bone 2008). The breadth of topics covered by these papers testifies to the diversity of issues that is currently of importance in seabird

research and the approaches available to confront such issues. However, this wide variation in topics and the inclusion of several valuable foundation descriptive studies hints at the enormous challenge facing Austral seabird research.

This first paper in the collection aims to draw out the cross-linkages between the contributions and poses the question: what next for Austral seabird research? But first we assert the importance of understanding seabirds and consider the conservation threats they face. Given the practical difficulties in studying these wide-ranging and long-lived vertebrates exploiting patchy and variable environments, why should Australian and New Zealand biologists bother with seabird research at all?

## THE IMPORTANCE OF AUSTRAL SEABIRD RESEARCH

Approximately one quarter of the world's seabird species breed in New Zealand and Australian territory (Taylor 2000). Eighty-four seabird species currently breed in New Zealand, of which 35 (42%) are endemic, breeding nowhere else (Taylor 2000). Australia serves as a breeding ground for 68 seabird species, of which 15 (22%) are endemic (Pizzey & Knight 2007).

The conservation status of seabirds is diverse and paradoxical; they exhibit a global distribution, population and biomass indicating that they are one of the most successful avian groups (Warham 1996), while at the same time being among the most threatened (Bolton *et al.* 2004). Almost half (59) of the world's 130 procellariid species, the largest grouping of seabirds, is classified as Critical, Endangered or Vulnerable according to IUCN criteria (IUCN 2007). Forty-seven of the 94 seabird taxa found in New Zealand are currently classified as Vulnerable or worse, with an additional four taxa lacking sufficient data for classification (Taylor 2000). Thirty-seven seabird taxa are so listed in Australia (Environment Australia 2008).

Many seabirds breed in densely populated colonies. Their density and abundance contribute to their role as a "keystone" and "ecosystem engineer" species on many islands, exhibiting wide-ranging effects on species composition, soil nutrification, aeration, and plant regeneration (Payton *et al.* 2002, Bancoft *et al.* 2008). The environmental modification caused by burrowing seabirds in particular can have a profound influence on island ecosystems. Wedge-tailed Shearwaters, *Puffinus pacificus* (J.F. Gmelin, 1789), for example, play a fundamental role in determining the species composition of colony associates on Rottnest Island (Bancroft *et al.* 2008).

As top predators, seabirds afford a unique opportunity for identifying and examining changes at lower trophic levels of the marine ecosystem (Furness & Camphuysen 1997). Coupled with their wide foraging range and, in some cases, trans-equatorial migration, seabirds are well suited to act as generalised biomonitors of marine ecosystem health. Changes in seabird populations, demographics and breeding characteristics can provide information about climatic and oceanographic changes, as well as marine pollutants (Furness & Camphuysen 1997). Seabird biomonitoring may also provide valuable information relevant to commercially important fish stocks which serve as prey for many seabird species. However, use of seabirds as biomonitors in this way relies on quantification of the impacts of land-based threats such as harvesting by humans (Kitson & Moller 2008, Lyver *et al.* 2008, Newman *et al.* 2008 a, b) and artificial mortality impacts at sea like fishing bycatch (Uhlmann *et al.* 2005, Waugh *et al.* 2008) so that background ecosystem effects can be reliably detected.

Commercial fisheries are of enormous economic importance for Australia and New Zealand. Seafood is New Zealand's fifth largest export, earning NZ$1.25 billion per year and supporting more than 22 000 jobs domestically (Seafood Industry Council 2008) In 2006/7 Australia's seafood exports totalled AUS$1.49 billion. The commercial fishing industry is the fifth most valuable Australian rural industry (Australian Department of Foreign Affairs and Trade 2008). Knowledge of the impact of fisheries on seabird populations is crucial for sustainably managing their coexistence. The studies included in this issue by Taylor (2008) and Waugh *et al.* (2008) examining feeding behaviour, and the extent of fisheries bycatch respectively, are important contributions towards this end.

The close relationship between seabirds and human societies has a long history. Until very recently, the harvest of seabirds and their eggs was an important source of sustenance for coastal human populations in most parts of the world (Moller 2006). Seabird harvests continue to have immense cultural importance for indigenous communities in both New Zealand and Australia (Skira 1995). The ongoing significance of the Grey-faced Petrel, *Pterodroma macroptera gouldi* (Hutton, 1869), and Sooty Shearwater, *Puffinus griseus* (J.F. Gmelin, 1789), (collectively known as "Tītī" or "Muttonbird") harvests for Māori in the North and South Islands of New Zealand is described by the contributions from Kitson & Moller (2008) and Lyver *et al.* (2008). An understanding of indigenous environmentality coupled with accurate population and harvest estimation is crucial to assessing and ensuring the sustainability of continuing harvests (Newman *et al.* 2008b). Maintenance of human cultures and local communities, their traditional lifeways, and Traditional Ecological Knowledge are now recognised as effective components of conservation efforts in Australasia (Wilson *et al.* 1992, Moller 1996, Skira 1996, New Zealand Conservation Authority 1997) and internationally (Borrini-Feyerabend 1996, Posey 1996, IUCN 1997).

## SEABIRD CONSERVATION CHALLENGES

The wide variation in the conservation status of different seabird taxa highlights the importance of measuring population changes to determine appropriate management strategies for each species. Unfortunately many threats potentially contribute to substantial population declines of some Australasian seabirds (Robertson & Gales 1998). Densely populated breeding colonies are particularly vulnerable to introduced mammalian predators, especially if adult birds are killed (Hamilton & Moller 1995, Jones 2000). Substantial efforts have been made to eradicate pests from breeding colonies in both New Zealand and Australia, and monitoring the response of seabird populations is of ongoing importance (Brothers & Bone 2008). Loss of habitat due to expanding urban development is concerning, although, as indicated by Rosier and Dyer's Heron Island study, some seabirds may be able to employ coping strategies to maintain breeding populations (Rosier & Dyer 2008). Detrimental human impacts on seabirds are also manifested in marine pollution. Hutton *et al.* (2008) reveal the disturbing extent of plastic ingestion in two shearwater species breeding on

Lord Howe Island. Falling ocean productivity and climate change impacts are suspected though not proven contributors to seabird decline (Scott *et al.* 2008).

The wide geographic distribution and long-range migration of seabirds may add difficulties to continuous study, but their philopatric behaviour, typically returning yearly to breeding colonies, makes annual monitoring of many species during the breeding season feasible on a large scale. For decades detailed research of the burrowing petrels was extremely difficult and studies of the behavior of seabirds while foraging and migrating were rudimentary, but this situation has improved with innovations in technology. The advent of data-loggers and telemetry (Shaffer *et al.* 2006, MacLeod *et al.* 2008) and use of sophisticated electronic equipment to detect the contents of nesting burrows (Lyver *et al.* 1998) are just some of the technological advances providing valuable new insights.

A less readily solvable challenge is that many breeding islands are remote, and research is consequently very expensive. It is therefore not surprising that many of the contributions to this special issue of *Papers and Proceedings of the Royal Society of Tasmania* are descriptive and short-term in nature. A "descriptive bottleneck" hampers conservation management and research in New World countries like Australia and New Zealand where the biota are comparatively new and unstudied. The population base and funds available for research are limited and there is a high number of threatened species per unit area. The USA spends 76 times more tax revenue per threatened taxon than New Zealand (Perley *et al.* 2001). As a proxy measure for financial capacity to respond to conservation needs through research and management intervention, this illustrates the severity of the funding deficiency. The Austral predicament is particularly acute for seabirds where much of the research is still focused on first steps – finding out what we have to work with (Bancroft *et al.* 2008, Brothers & Ledingham 2008) followed by rapid and necessarily crude assessment of conservation status (Taylor 2000, Brothers & Ledingham 2008).

Seabird conservation faces an added challenge from the vast geographic range of many species, roaming over several national and international oceanic jurisdictions. MacLeod *et al.* (2008) calculate that about 45% of the home range of Grey-faced Petrels falls in the international High Seas jurisdictional zone, with the remainder split roughly equally within Australian and New Zealand territorial waters. Oka (2008) examined Short-tailed Shearwaters before and after the trans-equatorial migration from Tasmania to the North Pacific and Bering Sea off the coast of Japan, a voyage which crosses numerous national jurisdictions. This wide-ranging journey across political boundaries emphasises a need for co-operation and data sharing to support neighbouring and sometimes distant nations to reach mutually important conservation goals.

## THE NEED FOR LONG-TERM STUDIES OF SEABIRDS

While several papers in this issue advocate more detailed studies on their specific research question (Powell *et al.* 2008, Taylor 2008), the most prevalent plea is for more long-term foundational monitoring data to be collected (Bancroft 2008, Hutton *et al.* 2008, Rosier & Dyer 2008, Scott *et al.* 2008, Waugh *et al.* 2008). It is only when these fundamental data are available that more advanced methodologies such as modelling of populations, reproductive success and density dependence can be applied. There is a clear recognition among Australasian researchers that long-term baseline data are gravely needed for large-scale questions to be addressed in seabird ecology, and for anthropogenic and environmental impacts to be properly understood.

The value of long-term studies in ecology is widely understood (e.g., Likens 1989), but the slow population turnover of marine birds, with their delayed breeding, slow rate of reproduction, long life-span and high adult survival rates makes long-term research and population monitoring programs especially pertinent (Wooller *et al.* 1992, Rosier & Dyer 2008). Factors affecting seabird populations are highly changeable year to year, with marked variation in reproductive success, chick and adult survival rates (Wooller *et al.* 1992, Newman *et al.* 2008 a, b, in press). Rare events, slow processes and subtle and complex phenomena are characteristics of a diverse and variable marine environment that can only be understood by long-term studies based on regular monitoring

Past demographic events can have persistent impacts on population abundance for decades to come, especially when multiple stressors are acting. For example, bycatch of Sooty and Short-tailed shearwaters in Korean and Japanese driftnet fisheries was particularly intense in the 1980s and banned in 1991 (Uhlmann *et al.* 2005). Computer simulations demonstrate that this bycatch has significantly depressed population abundance 15 years after its cessation. This reduction is likely to remain evident for a decade to come, with the length of recovery compounded by El Nino climate oscillations (Newman *et al.* 2008a). Disentangling the relative importance of different conservation threats that are sporadic and operate at vast distances from breeding colonies is difficult even if long-term and detailed population ecology studies are available. For most seabird species such research simply has not been done.

## WHY ARE CONSERVATION AND LONG-TERM STUDIES BEING NEGLECTED?

There is a severe shortage of funds for research in Australasia generally. Gross Expenditure on Research and Development (GERD) in New Zealand is around 1.2% compared to an EU-15 average of about 1.8% and target of 3% (Adams 2008). Already the United States is spending 2.67% of GDP on Research & Development (R&D), while Japan spends 3.06%. By comparison, Australia spent only 1.62% of GDP on R&D in 2002–03 (Australian Vice Chancellors' Committee 2005). In addition, little direct commercial stake in seabird research means that very little industry funding can be accessed by researchers. The high expense associated with research involving remote islands and sophisticated equipment such as telemetry and data-logging makes it particularly difficult to obtain investment in a 15-year lead time in long-term population studies before even basic descriptions of seabird population parameters can be described.

In both New Zealand and Australia, government research funding in recent years has increasingly moved towards output-based assessment measures to guide research investment. This shift towards a more competitive and adversarial funding scheme is likely to have significant consequences for seabird research. It will drive several changes in the nature and scope of research produced by

our tertiary educational institutions. Some of these changes are likely to be positive, but others may have unintended and unfortunate consequences for seabird research and conservation in particular.

One major consequence of output-based assessment models is the pressure placed on researchers to undertake short-term studies at the expense of good-quality long-term research (Adams 2008). Critics of the British Research Assessment Exercise (RAE) on which the Australian and New Zealand schemes are based argue that the biggest threat is "short termism" (Elton 2000). Elton (2000) specifically warns against other countries following the same path as the United Kingdom, yet New Zealand established a similar system two years later and Australia soon after. A restrictive short-term funding framework that typically has a 3–6 year funding cycle severely hinders the ability of seabird research to answer the large-scale questions that are most important in ecology.

The recent and ongoing publications of research from Fisher Island testify to the value and international rarity of long-term seabird studies. Once databases are established the long-term nature of the information becomes an enormous asset that facilitates research publication – the problem is initiation of the study and its maintenance through the 20–30 years before its long-term quality is reason enough to keep it going. Our main concern is that establishing new studies like that on Fisher Island will be increasingly difficult in the changing political research environment in Australia and New Zealand.

Under the present output-based schemes, research funding is determined in part by the number and "quality" of research publications. Publication in higher-ranked journals leads to more research funding, and because journal rankings are based on readership and citation rate, international journals invariably tend to be ranked more highly than local journals. While international research publication is important, the unfortunate consequence is that local and applied research more suitable for publication in locally-oriented journals is, in effect, financially discouraged. This is particularly negative for seabird research, which is often regionally situated and involves applied conservation and management.

## PUBLICATION OF APPLIED RESEARCH: A NEED FOR LOCAL JOURNALS

Local journals perform an invaluable function in promoting and disseminating research, particularly that which may be region-specific or provide baseline information, exemplified by the papers in this collection. Such studies are extremely valuable for Austral seabird research and management. Local journals enable the publication of locally-grounded, detailed research articles, some of which may be short-term, descriptive, or small-scale in nature. They are also invaluable destinations for taxonomic works requiring fast publication.

Furthermore, publication in local journals has a number of advantages over other journals. The time between submission and publication in such journals is usually much shorter, reaching the audience much sooner. In addition, contributions are typically reviewed by experts with a high level of local knowledge, who can provide detailed constructive feedback. The editors of local journals typically have a more direct interaction with authors, benefiting the final publication.

Despite their valuable roles and benefits, local journals are struggling to survive. Many are published by scientific societies with minimal financial resources, relying heavily on the goodwill of volunteers. We are indebted to Mrs Suzanne Skira and the Tasmanian Department of Primary Industries and Water for their generous grants from which the current issue of *Papers and Proceedings of the Royal Society of Tasmania* has been funded entirely. This issue will be followed by a second, comprising 12 contributed papers to be published on 28 November 2008, and funded by the Society.

Local journals are vital to accelerate the Austral seabird research past its predominantly descriptive phase and to prevent the unnecessary replication of research that has been left unpublished. We therefore salute the ongoing support of members of the Royal Society of Tasmania, and similar local science organisations, for making regular publications like this one possible.

## THE FISHER ISLAND STUDY: PATIENCE, PEOPLE AND RESEARCH POLITICS

Viewed in the context of increasing challenges to mount long-term seabird studies and publish the results, the value of the Fisher Island Short-tailed Shearwater study and Dr Skira's 25-year commitment to it (Ryan 2008) is outstanding. Why then was the Fisher Island study initiated and what were the key factors that allowed its persistence?

Short-term ecological studies, particularly those funded on the basis of competitive grants, are built around limited and specific key hypotheses. In long-term studies, however, it is often not appropriate to constrain data collection around too limited a set of specific objectives. The significance of the original questions will undoubtedly change throughout the life-time of the study, as new concepts are developed (Bradley *et al.* 1991). Long-term studies should not be unstructured and must be carefully designed to achieve a balance between data acquired to meet a specific hypothesis and data collected as general information. Cost-effective long-term studies need to allow the testing of retrospective hypotheses, and those undertaking them should be aware of the value of *a posteriori* data analysis.

Early in the Fisher Island study, Serventy (1957) stated that "The Fisher Island marking program is an aid to the study of all aspects of the natural history of the species". The efficacy of his generalised approach has become clear in hindsight. For example, data from the study have contributed to our understanding of the processes underlying life-time reproductive success (Wooller *et al.* 1989), the relationship between breeding success and subsequent survival (Bradley *et al.* 1989), philopatry (the outcome of a translocation experiment begun in the 1960s with results realised some 20 years later; Serventy *et al.* 1989), the inter-relationships between breeding success, breeding experience and mate fidelity (Wooller *et al.* 1990, Bradley *et al.* 1990, Bradley *et al.* 1995), the defining characteristics of the restricted number of breeders that contribute to the majority of the succeeding generation (Meathrel *et al.* 1993), prospecting and recruitment (Bradley *et al.* 1999), and the implications of breeding birds omitting breeding seasons (Bradley *et al.* 2000).

The major thrust of all of this work follows closely Dom Serventy's original concept of a fundamental understanding of the population biology of the species, informed through the lens of Life-History Strategy theory. With Dom's retirement,

the CSIRO wildlife laboratory near Perth involved Dr Ron Wooller, a seabird researcher from Murdoch University, who trained on the kittiwake study of John Coulson, one of the UKs leading long-term seabird studies. Consequently, with Irynej Skira and Ron Wooller in place, the Fisher Island study achieved a transition to a new scientific team, who were able to realise the potential of these data in shaping views of the life-history evolution and population dynamics of long-lived vertebrates.

Fisher Island researchers have always been cognisant of the tension of both how atypical the colony might be considered, and how its logistic convenience makes feasible the types of study outlined above. As a consequence, in 1991 Dr Catherine Meathrel, first involved in the project as a postdoctoral fellow, established a replicate study site on Great Dog Island as a monitoring and experimental site, to add to the information from Fisher Island and to provide some replication (Bradley & Meathrel 2006).

The Fisher Island research program is ongoing and now in the good leadership hands of Dr Meathrel. It is important that it continues as an inspiration for other teams to establish long-term seabird research programs. The worsening research climate for such studies is worrying at a time when the need for an understanding of global climatic and oceanographic changes and human impacts on biodiversity is becoming ever more critical.

## AUSTRAL SEABIRD RESEARCH: WHERE HAVE WE GOT TO AND WHERE DO WE NEED TO GO?

Given the wide range of challenges facing Austral seabird research and its relative infancy, it is perhaps unsurprising that a pervasive theme running throughout the papers in this issue is an appeal for future research; 11 of the 13 primary research papers in this volume explicitly call for further work to extend and clarify the foundational data reported here.

The advent of sophisticated telemetry and data-loggers now opens the door to understanding seabird behaviour at sea. While we share the excitement of filling in those gaps, in the longer run the critical research remains on understanding long-term population dynamics and what affects numbers. Otherwise we run the risk of locking into another raft of basic descriptive studies of increasing detail without confronting the crucial question for conservation management – what determines seabird population abundance and distribution?

Repeated and systematic monitoring is needed to make the most of the baseline estimates provided in this collection of studies and detect population declines and recolonisations as populations are restored (Brothers & Bone 2008, Newman *et al.* 2008b). Challenges remain to make these monitoring programs more accurate. For example, more reliable long-term records of occupancy estimates rather than just breeding burrow entrance density are needed (Scott *et al.* 2008). Long-term monitoring of the threats themselves is also needed to complement population monitoring if putative causes for declines or restoration are to be learned. For example, Hutton *et al.* (2008) call for further studies to assess the long-term trends in plastic ingestion by Flesh-footed Shearwaters on Lord Howe Island. Similarly, Waugh *et al.* (2008) call for tracking of the evolution of bycatch levels across areas, target fisheries and years, to better focus mitigation efforts to reduce seabird bycatch.

Studies of immigration and emigration are important for seabirds, but present specific difficulties. In procellariiform reproduction, a small portion of fledglings breed away from their natal territory and it is these that have a potentially crucial role in metapopulation dynamics and local population regulation (Scott *et al.* 2008, Newman *et al.* 2008 a, b). In general, metapopulation connectivity has not been studied amongst seabirds and the emergence of more fine-grained genetic and isotopic study techniques is eagerly awaited to establish global species population assessments – the expense of seabird surveys on remote islands usually precludes complete coverage of the breeding range. For example, rigorous estimates of other Sooty Shearwater populations would be beneficial for establishing whether the declines observed on the Snares by Scott *et al.* (2008) are occurring at the same rate throughout the species range, including the large and practically unstudied breeding populations in southern Chile and Terra Del Fuego.

Climate change is mentioned by several authors as a threat (Brothers & Bone 2008, MacLeod *et al.* 2008, Scott *et al.* 2008) but so far little direct evidence of its impact on Austral seabirds has been published. We predict this will become increasingly imperative as climate change accelerates (Hennessy *et al.* 2007) and interacts with existing threats to create synergistic effects from multiple stressors. In view of the importance of the oceans in producing oxygen and absorbing carbon dioxide, marine ecosystem health is paramount for the biosphere and the value of seabirds as biomonitors will be all the more evident.

Future studies must cross-reference behavioural and ecological patterns with spatial and temporal variation in potential conservation threats and spatio-temporal variation in oceanic conditions. This complexity is well illustrated by MacLeod *et al.* (2008)'s calls for future work to investigate the extent of overlap between the at-sea distribution of Grey-faced Petrels and commercial fisheries activity to determine the risk that this industry poses for this species. Environmental factors influencing the at-sea distribution of Grey-faced Petrels are the key to better understanding of the potential impact of climate change. Without this information, resource managers are unable to assess (1) direct and indirect effects of commercial fisheries, or (2) the effect of oceanic and atmospheric climate on Grey-faced Petrel population dynamics.

The dearth of long-term data sets cripples overarching understanding of patterns and trends in seabird population dynamics. The research in the present volume, in some cases comprising basic species surveys, points to a lack of foundational population data in Austral seabird research. Perhaps unsurprisingly, the current volume does not contain any experimental or manipulative studies, an absence that limits the scope of causal inferences and conclusions that can be drawn. Complete long-term baseline data must be built on by a mixture of comparative, experimental and modelling studies to complement the types of descriptive studies reported here (Hutton *et al.* 2008). Long-term descriptive monitoring studies form the basis of hypotheses for manipulative and experimental research designs, and can act as an external check on modelling studies (see Moller 2006). Ongoing monitoring can also reveal unexpected trends, generating hypotheses that would not emerge without continuing data collection (Burt 1994). In turn, demographic and population modelling can overcome some of the limitations of monitoring by allowing the prediction of responses to environmental changes. All of

these approaches require a solid foundation of long-term population and demographic monitoring data. This is where research effort must be focused, and where funding needs to be directed for an integrated and coherent research framework to emerge.

A further potential research avenue for learning how to conserve seabirds is the use of adaptive management. As a practical alternative to field experimentation, adaptive management theoretically offers longer term perturbations on more ecologically realistic spatial and temporal scales (Raffaelli & Moller 2000). For seabird ecology, this approach is most likely to be useful where management experiments involve removing threats at breeding colonies. Learning from adaptive management requires a Before-After-Control-Impact monitoring design to rigorously detect how a seabird population responds to mitigation of a threat, which in turn depends on standardisation and repeatability of survey methodology. While monitoring may be disparaged by some as simple, low-grade science, the long-term data sets it can produce are a necessity for more complex research methodologies to build on (Burt 1994). Many of the baseline studies reported in this volume will provide valuable benchmarks for future comparisons, just as Scott *et al* (2008) and Newman *et al.* (2008a) have been able to use fragmentary historical data to detect major changes in abundance. However, a more structured "active adaptive management" approach (Walters & Holling 1990) using multiple treatments, reference comparisons and adequate replication will be needed to determine and partition the causes of observed changes. Knowing why change is triggered (or why management intervention failed to trigger the expected gains) is the key to more empowered seabird conservation investments to stem widespread declines of Australia and New Zealand's diverse seabird species and to maintain ecosystem health in our magnificent seabird breeding islands. When coupled with a critical lack of funds for conservation management itself, the comparative lack of published locally-focused research and adaptive management to help prioritise management interventions becomes all the more risky for seabird conservation.

We urge more inclusion of ethno-ornithology to hasten research and guide conservation management. As Lyver *et al.* (2008) point out, indigenous peoples possess extensive knowledge of both harvested and non-harvested species, and this knowledge can be regionally unique. Indigenous knowledge bases must not be devalued for their regional diversity and specificity. There must be an appropriate outlet for dissemination and its importance should be reflected in funding frameworks. Indigenous perspectives on environmentalism have particular relevance to regional conservation efforts, and the narrow Western-science-oriented framework on which the Australian and New Zealand funding schemes are based potentially hinders the valuable participation of indigenous peoples in the decision-making process of environmental management initiatives.

Despite the immense challenges facing the field of Austral seabird research, progress is certainly being made and obstacles overcome due to the determination and perseverance of researchers. While the current trend towards increasingly contested research funding in Australia and New Zealand threatens progress, potential benefits of a more rigorous research framework may emerge. We must strive to ensure that the long-term studies currently operating continue, and work hard to generate an environment in which new long-term studies can be established. From here, a research framework combining advanced and complementary methodologies can develop and continue to address the crucial overarching questions in seabird ecology in this globally important region.

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# IRYNEJ SKIRA (1950–2005): A SCIENTIFIC LIFE

by Lyndall Ryan

(with 15 plates)

**Ryan, L.** 2008 (31:x): Irynej Skira (1950–2005): A scientific life. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 9–20. ISSN 0080-4703. School of Humanities and Social Sciences, Faculty of Education & Arts, Ourimbah Campus, University of Newcastle, PO Box 127, Ourimbah, NSW 2258, Australia. Email: Lyndall.Ryan@newcastle.edu.au

The unexpected death of wildlife biologist, Dr Irynej Skira, at the age of 54 in February 2005, sent shockwaves through the scientific community. His longitudinal study of Short-tailed Shearwaters, *Puffinus tenuirostris*, also known as muttonbirds, was about to receive international recognition, while his recent second marriage appeared to open up a new chapter in his personal life. Who was this remarkable man, what was his contribution to science and what is his legacy? This article draws upon the funeral notes and memories of his family, friends and colleagues, to provide an overview of his personal and professional life. It shows how his two periods of fieldwork on Macquarie Island developed his interest in wildlife biology and how he extended it to assume responsibility for the longitudinal study of shearwaters. It explores his particular contribution to the study over nearly three decades and then discusses his wider contribution to conservation and preservation of wildlife in Tasmania. The article concludes with an assessment of his legacy.

**Key Words: Irynej Skira, Short-tailed Shearwaters, *Puffinus tenuirostris*, muttonbirds, longitudinal studies, Tasmania, Macquarie Island, Fisher Island, The Snares, conservation.**

## INTRODUCTION

At the end of 2004, Irynej Skira had reached a turning point in his life. He had just married his second wife, Suzanne Frances Edwards, at a ceremony on his beloved Fisher Island. He had accepted an invitation to visit The Snares, the seabird sanctuary south of New Zealand and thus fulfill a long-held dream, and he had been invited to present a paper at the Second International Manx Shearwater conference in Belfast in Northern Ireland in August 2005. This was long-awaited international recognition that he was the world's foremost researcher on Short-tailed Shearwaters, *Puffinus tenuirostris* (Temminck, 1835) (called "muttonbirds" or "yollas" in Tasmania).

But he would not live to enjoy any of them. A month after his marriage to Suzanne, he set off for The Snares (pl. 1). An unsuspected heart infection erupted, leading to a stroke. He was airlifted to Dunedin by helicopter and flown back to Hobart a few days later. He died in the Royal Hobart Hospital on 18 February 2005 at the age of 54.

Irynej's unexpected death sent shockwaves through the scientific community. At his funeral, in St Joseph's Catholic Church in Macquarie Street, Hobart, just up the street from his workplace, his long-time friend and colleague Nick Mooney told an overflowing congregation of grieving family, friends and workmates that Irynej had seemed indestructible. He had conducted his research into muttonbirds (pl. 2) for more than 25 years; he had weathered endless departmental restructures; he was a gentleman who valued politeness and scientific integrity; and his quiet professionalism gave him the aura of rock-like stability in "a sea of nonsense". His loss was irreparable (Mooney 2005).

Who was this remarkable man? What was his contribution to science? What legacy has he left us? This article attempts some answers to these questions.

*PLATE 1*
*Irynej Skira on The Snares, January 2005.*
*(Photograph: Lisa Daglish)*

## CHILDHOOD 1950–71

Irynej Osyp (Joseph) Skira was born at the Queen Victoria Hospital in Launceston, Tasmania, on 18 May 1950, the second son and middle child of Joseph and Irena Skira, who had arrived in Tasmania as refugees from Ukraine in late 1948. Born on the feastday of St Irene in the Ukrainian Catholic Church calendar, Irynej was named after St Irenaeus, a scholar and saint from the second century AD. According to his sister, Eva, his unusual first name (pronounced *i-rin-ay*)— which no one could say, spell or write in the then-conservative Anglo-Celtic culture — shaped his unique character from the outset (Skira, E. 2005).

Irynej told me that his father, Joseph, who had a background in engineering, and his mother, who had started a medical degree in Europe, had been given two choices of residence in Australia: Queensland or Tasmania. They chose the latter because of its cooler climate. In Irynej's view his family made the right choice. He was the first of their children to be born in Tasmania and from his early days had an empathy with his island home. He rarely felt the cold and appeared to be genetically attuned to the vagaries of the Tasmanian climate.

Joseph was employed as a foreman in a local saw-mill, initially out in the yard and maintaining the machinery, subsequently in an administrative position, while Irena took on part-time work as a nursing carer. They built a modest home at St Leonards, 12 km southeast of Launceston, in the shadow of Mount Barrow, and the family moved there in 1954. Both Joseph and Irena contributed widely to their community and became actively involved in school, church and Ukrainian associations, as well as the Launceston branch of the Good Neighbour Council.

Like so many postwar migrants with backgrounds in the professions, but unable to gain employment in their respective disciplines, Irynej's parents pinned their hopes on their children to achieve professional success by working very hard, valuing excellence in academic studies and expecting their children to have the same ethos; these values became a defining aspect of Irynej's character.

As a child, Irynej was shy and had a stammer that would remain with him all his life. In the 1980s Irynej had some speech therapy and his colleagues immediately bore the brunt of the new confidence through wit and cheek, at first putting it down to a new relationship (Nick Mooney pers. comm.). But he also had a bright, enquiring child's mind, and his mother remembers him as the most restless and exploring of her three children. Irynej later recalled finding native animals outside the back fence, building farm fences in the backyard, taking long bike rides into the mountains behind his home and joining the scouting movement, which fostered in him a love of bushwalking throughout Tasmania. At first he wanted to become a farmer, but after a while animal husbandry was replaced with a general love of nature and wildlife, in particular birds. His passionate relationship with the natural world was already well in train at this time.

He attended St Leonards Primary school and then St Patricks College in Launceston, from where he matriculated in 1967, excelling in Latin, mathematics and science. He also excelled at sport. He loved cricket and Australian Rules Football, although he did break his bones a couple of times, and he was a good runner and a good swimmer. He also learned the piano, but later said he was relieved when his music teacher suddenly died, and he no longer had to take lessons. His parents, however, had given him a deep love of classical music that would become an important source of enjoyment in adulthood.

Irynej also loved books. His sister Eva remembers that at home he would spend hours looking at reference books and encyclopedias that had both words and pictures. In his early adult years he began to collect some fine books, many cataloguing birds (Skira, E. 2005). This passion for collecting continued for the rest of his life. He also had an eye for black and white photography, drawing and sketching, later becoming interested in black and white prints.

In his later high school years, he worked in the long vacation in a saw-mill, where the incessant loud noise of the electric saws permanently deafened him in one ear — a not uncommon injury in an industry that, in those days, had no occupational health and safety regulations. Partial deafness became another factor in his defining status, together with partial colour blindness and short sight. None of these disabilities were of concern to Dr Bob Green, the curator of zoology at the Queen Victoria Museum, who took him on his first field trips and started him on the road to becoming a natural scientist.

In 1968, he went to the University of Tasmania to study for a Bachelor of Science degree, majoring in zoology, and moved to Hobart. At the end of his first year he gained a Commonwealth Scholarship to complete the rest of the degree. He excelled in fieldwork and bushwalked the southwest of Tasmania. By the time he had completed his undergraduate studies, there were few areas of his island home that he had not traversed on foot or by pushbike. He had already become a true Tasmanian.

## TASMANIAN PARKS AND WILDLIFE SERVICE AND MACQUARIE ISLAND 1972–77

On Irynej's twenty-second birthday, as a "chirpy new science graduate" he began employment with the Tasmanian Parks and Wildlife Service (TASPAWS) as a park ranger at Cradle Mountain National Park (pl. 3). His colleague Trevor Weston, who started on the same day, later pointed out that science graduates were quite a novelty then and "Iry became the department's most highly qualified cleaner" (Mooney 2005).

"Iry" had other ideas. He wanted to join the scientific section of Wildlife. By the end of 1972, he was one of two biologists on Macquarie Island where TASPAWS had taken over rabbit eradication work from the Department of Agriculture. One of his duties was to record the number of cats and their colour. Many years later, it was found that Irynej's sample of cat colours did not match those of other researchers, ginger being under-represented. Everyone had forgotten his partial colour blindness (Mooney 2006).

According to Nick Mooney, Irynej came back from Macquarie Island inspired by curiosity-led research and, after a short stint at Lake St Clair National Park, returned to Macquarie Island for 16 months from November 1973 to March 1975 (pls 4, 5). He started studying the rabbit population with a view to impact assessment and control, setting up a monitoring program that is still in operation 35 years later. This led to his first scientific publications and became the basis of his MSc degree, which was awarded in 1980. Irynej demonstrated that as an active scientist he would make a contribution to scholarly research.

*PLATE 2*
*Short-tailed Shearwaters,* Puffinus tenuirostris, *also known as "muttonbirds" or "yolla" in Tasmania.*

*PLATE 3*
*Irynej Skira in Cradle Mountain National Park as a ranger.*

*PLATE 4*
*Irynej Skira on Macquarie Island with a Brown Skua,* Stercorarius antarcticus.

*PLATE 5*
*Irynej Skira with a large male Elephant Seal,* Mirounga leonina, *on Macquarie Island.*

On his second return from Macquarie Island, he lobbied hard for scientific input into wildlife management and eventually managed to transfer from the Parks to the Wildlife Section as one of its first scientific officers. According to Nick Mooney, "No doubt island life brought seabirds to Iry's attention and what better seabird to study on his return than the Short-tailed Shearwater" (Mooney 2006: 41). He was indeed well-suited to study this bird, because they are identified by the shape of their tails and wing spans, rather than by their colour. His short-sightedness enabled him to notice details of their eggs and burrows and his partial deafness attuned his senses to their smell and sounds. In studying the muttonbird, Irynej turned all his perceived physical disabilities into positive, even necessary, attributes.

## THE SHEARWATER/MUTTONBIRD PROJECT 1977–95

In 1947, Dominic Serventy, one of the fathers of Australian ornithology, had started what would become 30 years' work on Short-tailed Shearwaters, a trans-migratory seabird that breeds only in southern Australia and migrates to the far northern hemisphere for the Austral winter. On their migration route the shearwater can fly over 16 000 km each way, which is nearly the same distance as to the Moon and back.

The shearwaters return to Australian waters, largely to islands in Bass Strait, from September and their breeding timetable is strict. It was once thought that they all laid their eggs on the same day, which is close to the truth: 85% of all eggs are laid within three days either side of 24–25 November. The eggs are incubated for 54 days and chicks fledge about 100 days after hatching.

Dominic Serventy was determined to study the population dynamics and breeding habits of these birds, and to make them the subject of a long-term study. In 1948 he built a hut on Fisher Island in Franklin Sound, just off Lady Barron on the southern side of Flinders Island, and made it the centre of his research (pls 6, 7) . Much of this 0.8 hectare island is bare rock, with the rest covered with *Poa* tussock grass and friable soil.

By the mid-1970s, Serventy was ready to retire and persuaded TASPAWS to take over responsibility for research

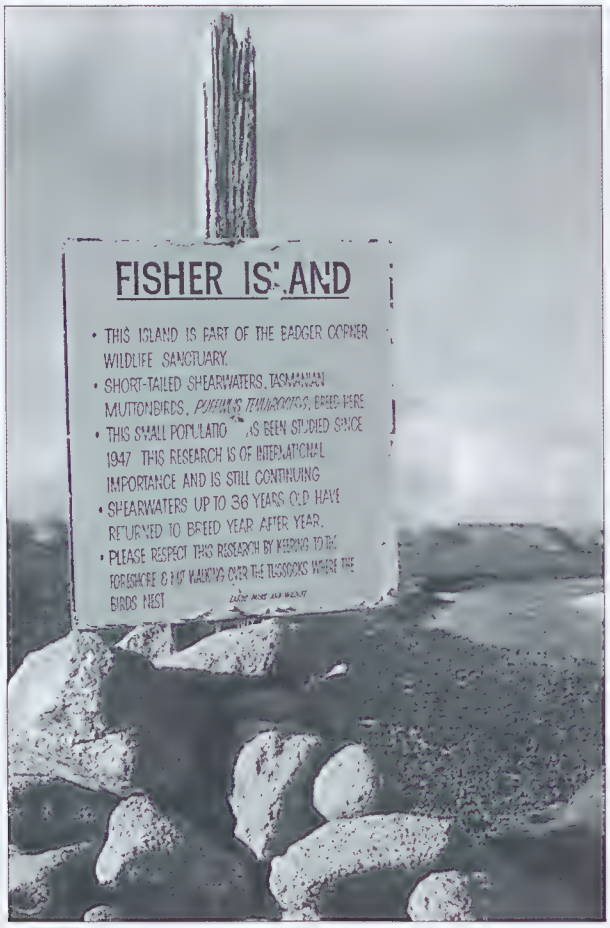
FISHER ISLAND
• THIS ISLAND IS PART OF THE BADGER CORNER WILDLIFE SANCTUARY.
• SHORT-TAILED SHEARWATERS, TASMANIAN MUTTONBIRDS, PUFFINUS TENUIROSTRIS, BREED HERE
• THIS SMALL POPULATIO ,AS BEEN STUDIED SINCE 1947. THIS RESEARCH IS OF INTERNATIONAL IMPORTANCE AND IS STILL CONTINUING
• SHEARWATERS UP TO 36 YEARS OLD HAVE RETURNED TO BREED YEAR AFTER YEAR.
• PLEASE RESPECT THIS RESEARCH BY KEEPING TO THE FORESHORE & NOT WALKING OVER THE TUSSOCKS WHERE THE BIRDS NEST

*PLATE 6*
*Fisher Island off Lady Barron, Bass Strait home of the longitudinal study of Short-tailed Shearwaters.*

*PLATE 7*
*Irynej Skira offshore from Fisher Island in December 1992. Clearly visible on the island is the hut from which the Short-tailed Shearwater study was, and is, conducted.*

*PLATE 8*
*Irynej Skira on Fisher Island in March 1976 when working as a field assistant for Domenic Serventy.*

*PLATE 9*
*Many who worked on the Short-tailed Shearwater project photographed on nearby Green Island in December 1988. Back (standing) from left to right: Nick Parrott, Irynej Skira, Anna Navin. Front (seated) from left to right: Rod Leaman, John Cassoti, Georgina Davis, Stuart Bradley and Ron Wooller.*

and management of the program. At first Nigel Brothers kept the study going, but Irynej, who had worked as Serventy's field assistant or "dinghy boy" in 1973, impressed the famous man and was anointed by him to continue his work (pl. 8). He could see that Irynej had an empathy with the muttonbirds, the tenacity for a long-term study and the scientific temperament to carry it out. I think, too, that he was impressed by Irynej's family background, for Serventy's parents were also migrants from Eastern Europe, who valued scientific study and had given him the work ethic of sticking with a long-term project. Irynej assumed

formal responsibility for the program in December 1978. Six months later Dom wrote to Irynej: "you are now thoroughly immersed in the Fisher Island work. When I come over in November it will have to be as *your* assistant" (Skira 2005b: 4).

Around this time two personal events occurred. First, his father Joseph passed away in July 1977, which had a significant saddening effect on the whole family, particularly Irynej and his mother. Second, in December 1978 he married Peggy Briggs, who was 17 years his senior and worked in the TASPAWS Office in Magnet Court at Sandy Bay. Earlier, before his first stint on Macquarie Island he had a relationship with a young woman, with whom he had a daughter Sarah, but it didn't work out. He later told me he was "a wild young man back then", with a penchant for motor bikes, which he rode at breakneck speed up the Midland Highway. Marriage to Peggy gave him confidence and maturity.

By then he looked like everyone's idea of a wildlife biologist. He had a lean frame and dark, good looks, with large brown eyes, longish, dark curly hair and a beard, and wore glasses. He was more at home in the TASPAWS field uniform of dark green trousers, shirt and jacket than an office suit, and clearly preferred the uncertainties of fieldwork to the more predictable life of the office. As a settled married man he was well-suited to continue what would become the world's longest longitudinal study of any seabird in the world.

Fisher Island would become the central focus of his professional life for the next 25 years and the jumping-off point for several important projects, scientific and non-scientific (pl. 9). It is extraordinary that this small island would be the centre of a rich scientific life that opened up new worlds for Irynej in so many different directions. As Suzanne Skira later wrote:

> *Research on Fisher Island forms the cornerstone of all management decisions on the species. Here Irynej observed many aspects of shearwater life, including population dynamics, survival, breeding and feeding. One of the strengths of the Fisher Island study is its simplicity of monitoring. Approximately 80 pairs breed on the island each year in three discreet colonies: Home Rookery, South Rookery and Potts Point. All the burrows are mapped and have their own pole with an identification number. Irynej determined burrow occupancy by using a stick 60 cm long to feel inside the burrow. If it was occupied he gently removed the bird, recorded the sex and band number and checked for an egg. New birds were banded and old bands replaced.* (Skira 2005b: 4–5)

Irynej led fieldtrips to Fisher Island three times a year: in December to count the eggs, in March to count and band the chicks and in April to monitor the muttonbird "harvest" on other islands in Bass Strait. It was arduous work, but Irynej, undaunted, extended Dom Serventy's work of banding the birds by monitoring them in their burrows with transects (pl. 10), and with blood and feather sampling as well as weighing and measuring. According to Nick Mooney, Fisher Island also provided first-class field experience for new employees in TASPAWS over the next 28 years.

> *The benefit of the shearwater experience and a stint on Fisher Island became a rite of passage for several generations of local scientists, rangers, students and naturalists. He introduced TASPAWS rangers to the eccentricities of scientists and the cooperation and discipline needed for long-term scientific research."* (Mooney 2006: 41)

*PLATE 10*
*Original transects on Fisher Island were laid down by Dominic Serventy with these same poles. Here they are monitored by Irynej Skira (centre) in 1995.*

In those days, Occupational Health and Safety guidelines were "a matter of self reliance and common sense" and mobile phones had not yet been invented. When contact was necessary, the ranger on Flinders Island would sometimes fire a gun from the shore to try to get Irynej's attention. It was only when Irynej got an outboard motor in the mid-1980s that there was an accident. A senior ranger (no less than Peter Mooney, brother of Nick Mooney) fell out of the dinghy and had to tread water for over an hour until the circling dinghy ran out of petrol, before he could rescue himself. It is testimony to Irynej's great care and preparation that more serious accidents did not occur.

Nick Mooney later pointed out:

> *Many of us met our first muttonbirds on Fisher Island. The early morning and late evening stints, arms buried in shearwater burrows, the battle of wits with the local rats, Iry's bashing of abalone, and the endless rowing of a battered dinghy are all etched in our minds."* (Mooney 2006: 41)

In this way he trained many generations of young and not so young scientists in the art of recording and banding the birds. He also trained them in thorough preparation for fieldwork, an essential aspect of the life of a field biologist. This included the ordering of food supplies and scientific equipment and how to account for everything on the tightest possible budget. His field notebooks, which contain his distinctive "scientific" style of handwriting and recording observations, stand as a model of scientific note-taking, while his black and white photographs are testament to his keen eye for detail.

Over the next two decades, Irynej published more than 40 scientific papers on the population dynamics of the shearwaters and other seabirds. He found that their population is about 23 million adults and that they first breed between four and five years of age and can live up to 38 years. He also found that there are 250 muttonbird colonies in Australia, of which over 200 are in Tasmania, comprising 80% of the total population. The colonies are distributed around the coast of Tasmania and its offshore islands. The largest colony is on Babel Island, with over two million burrows. By the time of his death in 2005, more than 70% of the breeding colonies were protected

on reserves.

In 1981 he gained a grant from the Australia–Japan Foundation to study seabirds and Japanese Snipe, *Gallinago hardwickii* (J.E. Gray, 1831), in Japan, and in 1985 gained a further grant from the Yamashina Institute of Ornithology in Japan to follow the seasonal patterns of migration of the muttonbird, from Bass Strait to the Bering Strait, the west coast of Canada and across the Pacific Ocean back to Bass Strait. To this end he made a six-week return voyage on a cargo ship from Japan to the west coast of the United States via Hawaii.

This extraordinary journey brought home to him the international and regional significance of the shearwaters and how they were an important marker of climate change. He published several papers that noted increases in ocean temperatures in various parts of the Pacific Ocean in different seasons and the effects on the muttonbird's migratory habits. A paper he published in 2003, for example, indicated that the increase in winter ocean temperatures off the east coast of New South Wales in 1999 had resulted in there being insufficient food for the muttonbirds, which had led to a significant increase in the death rate, far beyond the norm, and that a further increase in ocean temperatures in the following years had led to another increase in muttonbird deaths (Skira 2003b). By then scientists had identified El Nino and its effect on drought in Australia. Irynej's interest, however, was the impact on muttonbirds. Put together with Dom Serventy's work, the longitudinal study provided important information about the fluctuations in bird populations and in ocean temperatures over nearly 60 years.

As Nick Mooney pointed out:

> *These long-term studies are both rare and valuable: rare because they require dogged persistence and valuable because they show natural fluctuations in populations over meaningful periods of time and, thus, can also reveal abnormal change. Some of Iry's study birds were actually older than him — at one stage, one was at least 46 years old" (Mooney 2006: 41).*

Suzanne Skira noted that when she worked with Irynej on the island in 2003 that "the older pre-banded birds seemed to anticipate Irynej's handling and remained calm." (Skira 2005b: 5)

The legacy of this work is that Irynej produced the best knowledge in the world on the ecology of shearwaters, which has led to better protection and conservation of species. Above all he demonstrated the value of long-term monitoring programs.

## THE SHEARWATER PROJECT AND THE TASMANIAN ABORIGINAL COMMUNITY

One of the first groups of people Irynej met when he became responsible for the shearwater management program was the Aboriginal community, which had been engaged in seasonal muttonbirding since time immemorial and on a commercial basis since the early nineteenth century (pl. 11).

Two Aboriginal women, Marge Mansell and Phyllis Pitchford, showed him how to pluck muttonbirds and how to locate the best burrows. After Irynej had finished his Master's degree, he began to realise that the history of muttonbirding in relation to the Tasmanian Aborigines and other human groups needed serious study, so he enrolled as a doctoral candidate at the University of Tasmania. At his

*PLATE 11*
*Harvested muttonbirds in the Bass Strait islands.*

PhD ceremony dinner some years later, Irynej told me that he had great difficulty in finding a university department and a supervisor for the project. Eventually he was accepted by the Environmental Studies program in the Geography Department, where he was supervised by Peter Hay, who had enormous faith in Irynej and the project. He rapidly mastered the disciplinary demands of historical research, and set about the task with flair and confidence.

In the first section of the thesis, Irynej set out the ecology of the muttonbird, from its evolution, distribution and abundance, migration, feeding, breeding biology and mortality rates. This was based on his own scientific research over the previous decade. In the next section he outlined the exploitation of muttonbirds in the past by the Indigenous peoples in Tasmania and on the east coast of Australia, as well as in New Zealand and the northern hemisphere. For this information he drew on archaeological papers, reports and theses and revealed for the first time the importance of muttonbirds as a major source of food and ritual in these societies.

In the third section, he outlined the arrival of European explorers and sealers to Tasmania in the early nineteenth century and their interaction with Aboriginal people in northern Tasmania, which led to the development of the Islander community in Bass Strait in the later nineteenth century and their conflict with European farmers and settlers who took out leaseholds on the islands to graze sheep and cattle. He then explored social change in the Islander community in the first half of the twentieth century, in relation to the Cape Barren Island Reserve Acts of 1912 and 1951 and their impact on the muttonbird industry. The final section of the thesis explored the commercial aspects of the industry to the present (Skira 1993).

Irynej produced a bold and innovative thesis, one of the first in Australia to explore the relationship between food resources and human groups in the past and the present. The examiners were historians, a prominent environmental historian in one instance, and a historian of Aboriginal–European interactions on the other. The doctorate awarded in 1995 was the pinnacle of his career to that point. In Hay's opinion, its disciplinary sweep makes it an extraordinary study. It is a great tragedy that it has never been published.

## EXTENDING HORIZONS 1995–2005

By then Irynej's marriage to Peggy had ended and his older brother, Peter, had died, leaving a deep emotional scar. He moved to a cottage at Kingston Beach and was often seen in Hobart scouring bookshops for rare titles and taking coffee with friends at Fullers Bookshop. He shared an interest in antique colonial furniture with Ray Prince and in painting with George and Mary Davis. Over the next five years, he met a number of women with whom he developed very close relationships, but none could match his emotional depth and intensity. He would have to wait a few more years before he found the woman who did. In the meantime his close friends could see that he was unsettled and unfulfilled. He was sustained by exciting new developments in the study and management of muttonbirds.

In the late 1990s, he made contact with the Māori community in New Zealand to learn the ways they engaged with muttonbirding. In 1999 he hosted four New Zealand scientists on Fisher Island to "... look at the Tasmanian muttonbird harvest and ecological research" (Moller 2000: 4). Henrik Moller was

> *"... amazed (and envious) of the detailed scientific knowledge of the Short-tailed Shearwater. They have measurements of the size of pretty well every colony in all of Australia and one very long study of the birds on Fisher Island. It shows the value of long term research – those scientists over there have been banding and surveying their muttonbirds for 50 years!"* (Moller 2000: 4)

He also noted that Irynej had recorded their lives in ways that more than matched records of human activity. "One bird had outlived two lots of metal bands, and was last seen in 1994 at the age of 38 years" (Moller 2000: 6).

Irynej used their visit to initiate contact between the Maori community in New Zealand and the Aboriginal community in Tasmania so that they could visit each other during the muttonbird season and explore and understand the different ways they conducted the industry. He also obtained funding from the Contract Employment Programs for Aborigines in Nature and Cultural Resources Management to employ up to ten Tasmanian Aboriginal people to undertake research into muttonbirds from an Aboriginal perspective.

His enthusiasm for collaboration with scientific colleagues was a marked feature of his career. According to Nick Mooney, "his many co-authorships are testament to his extensive collaborations" (Mooney 2006: 41). He never worried overly about his ownership of data and always welcomed people to the field. For example, when I arrived in Tasmania in February 1995 to collect new material for the second edition of my book, *The Aboriginal Tasmanians,* he gave me access to his own unpublished doctoral thesis and read many drafts of my manuscript. I admired his ability to deal with pressure, manage ambiguity and change in dealing with the Aboriginal community over "cultural muttonbirding" and the way he balanced their claims against environmental, social and regulatory concerns, and Government policy. Here was a scientist who really cared about Aboriginal people and the muttonbirds and the place of both in a complex environment. He also helped students from mainland Australia and overseas universities with shearwater and gull studies, some on his beloved Fisher Island. He also shared his wealth of knowledge with local students, wildlife care groups and any interested volunteers.

This is the sign of a true scholar.

Irynej also encouraged artists and writers to visit Fisher Island and they often gave him some of the products of their creative endeavours. When I first met Irynej in 1995, he had a significant collection of drawings of plants and birds as well as paintings of Fisher Island and the surrounding islands in Bass Strait. He encouraged many friends to join him for a week or so at a time, to give them a break from their own busy lives, but in reality to assist him in recording and banding the birds. In 1997, he hosted a Fisher Island fiftieth birthday party in Hobart, inviting all those who had been there, to join him in the celebrations. The large turnout of people from a wide variety of backgrounds, including medicos, academics, artists and writers as well as his scientific colleagues, is testament to his wide circle of friends who were proud of their association with him, with Fisher Island and with the project.

Irynej's professional life as a wildlife biologist also continued to vary. For example, he helped with the small mammal surveys on Hunter Island and even joined Nick Mooney in search of thylacines after the famous 1982 report by Hans Naarding. Nick always found him good company, "suitably eccentric and quick to laugh, with a dry, incisive wit. We didn't always agree, but why would we want to?" (Mooney 2006: 41)

In 1993, he gained a significant grant from the Australian Nature Conservation Agency and National Estate to investigate the threat from the North Pacific seastar on native marine invertebrates and became a founding member of the Seastar National Taskforce.

After the *Iron Baron* oil spill in Bass Strait in 1996, Irynej took responsibility for organising the response equipment, and helped with seal surveys in Bass Strait. At other times he moved, to Bass Strait, errant seals which had attacked salmon farms in southern Tasmania. This involved capturing the seal and putting it in a saltwater tank in a cage on a trailer and then towing it up the Midland Highway for release at a beach at the mouth of the Tamar River. At one stage in 1996 and 1997, he was making four or five trips a week, a terrible waste of his scientific skills and abilities. Then he became part of a seal tagging program and assisted in tracking the remarkable behaviour of "Radar", an adolescent male seal which he captured several times at salmon farms and then released into the Tamar before it finally disappeared into the South Pacific Ocean. "Radar" aroused considerable media interest and Irynej was photographed at least once in the Hobart *Mercury,* with his boisterous charge in tow.

But Irynej was equally famous for not doing things that he thought were a waste of time or just plain silly. Every now and then he would apply for promotion to a senior position, only to be told that he should wear a suit and attend more meetings. As his colleague Nick Mooney recalls: "few of us escaped the odd sharp rebuff in this regard" (Mooney 2005: 3).

Among Irynej's more thankless tasks, was as secretary to the Department's Animal Ethics Committee, where he had to endure some aggressive lobbying and, at times, rather nasty criticism, which he weathered with complete professionalism. Other tasks included responsibility for government management of zoos and wildlife parks in Tasmania — a frustrating and difficult task for a wildlife biologist. However, he dealt with the managers of these outfits more successfully than his predecessors by adopting a firm but civilised approach and by refusing to be drawn into slanging matches. His exemplary professional behaviour as

government representative on the Antarctic Animal Ethics Committee also stood him in good stead.

Another aspect of his professional life was his membership of the Australian Bird Study Association. He contributed to the Seabird Island Series and was the driving force in the preparation of 15 of the subsequent 25 descriptions of Tasmanian Islands.

In the twenty-first century, recognising the value in understanding common species — perhaps those most important to ecosystems — Irynej became increasingly frustrated with the trend towards studying or conserving only things that were on the Endangered Species List. He also despaired of the destruction of Tasmania's old-growth forests, lamenting that protective protocols applied only to endangered species, guaranteeing the endless logging of timber.

In 2003, he disputed the demand by the Mayor of the West Coast that bag limits for commercial muttonbirding should be increased. He was always concerned about over-harvesting:

> *The limits are there for a reason. Only one third of chicks return to breed, laying just one egg per season after they reach the minimum of five years of age. Mortality rates are high with 10% of adults dying each year. Every muttonbird season more than a quarter of all burrows are badly damaged and vegetation destroyed, all to reach one bird. I've seen evidence of not only a spade being used to get to a bird, but where people have punched through the earth because they can't reach the end of the burrow. There are places where vegetation has been taken to with a chainsaw to obtain easier access.* (Skira 2003a)

He gained some of the first National Heritage Trust grants for feral species management and organised cat control on seabird islands in the Furneaux Group, and pig control on Flinders Island. He then helped with a biodiversity survey on a large midlands property and began to realise that it could be the centre of another long-term study. This determination to do practical things was typical of his nature.

It is fitting that Irynej would meet his soul-mate at a muttonbird colony. He always believed that he should educate the public about shearwaters and every summer he led public field tours at Clifton Bluff muttonbird colony near Hobart. There in 2000, he met Suzanne Edwards, who was a nurse. They had an instant rapport, and it was not long before they began to plan their future together. When they were married on Fisher Island in December 2004 (pl. 12), they both believed that their best years lay ahead. His friends were delighted that he had at last found personal happiness and fulfillment. One of them built a new house to the couple's specifications at Kingston Beach, which included a specially designed library for Irynej's collection of rare books, prints, stamp collections and antique colonial furniture. They expected to move in when Irynej returned from The Snares in February 2005.

## THE SNARES

Irynej always had an irresistible sense of mission to go to The Snares. Its extraordinary diversity of seabirds as well as its remoteness held an attraction that matched his passion for Macquarie Island. However, the opportunities to go there were few and far between. When Henrik Moller invited him to join an expedition with Chris Powell and Corey Bragg

*PLATE 12*
*Irynej and Suzanne Skira on East Kangaroo Island on 13 December 2004, prior to their wedding on Fisher Island. (Photograph: Dawn Zelman).*

from the University of Otago at Dunedin, he believed it was an important moment in his scientific career.

According to his wife, Suzanne, Irynej was feeling tired the day he left for Dunedin. But they put it down to the strain of preparation for this long-awaited trip and believed that once he got under way, he would return to "normal", a word he often used to describe the ordinary. Besides, he had always enjoyed extraordinarily good health and could barely recall a day's illness. When he arrived in Dunedin he told Suzanne in an email that he still felt tired, but that the excitement of almost being at The Snares had taken over.

He arrived on Friday, 21 January 2005 and had time to record in his diary every animal and bird he came across before his last entry three days later: "Nice calm morning. Blue skies. Got up approx 6 am — noisy shearwaters everywhere". By the end of the following day, he was fighting for his life and awaiting transport by emergency helicopter to Dunedin Hospital. He died in Hobart 25 days later.

The last diary entries struck a chord with his colleagues. According to Nick Mooney, they were:

> *... 100% Irynej — thorough and dry. He was obviously pleased to be there, in the best of company, both people and wildlife — sea lions, shearwaters and a black tom tit. I'm sure he wouldn't mind me saying he more or less died with his boots on."* (Mooney 2006: 41)

## INTERNATIONAL RECOGNITION

When news of Irynej's death reached the international scientific community, the Second International Manx Shearwater Workshop in Belfast was dedicated to his memory. There his wife Suzanne presented a paper to a hushed audience that understood, perhaps for the first time, his immense contribution to the field (Skira 2005b). Obituaries followed in several scientific publications and journals devoted to the study of birds (Moller 2005, Murray 2005, Sharpe 2005, Skira 2005c, Mooney 2006).

## LEGACY

Irynej Skira was a professional wildlife biologist with the Tasmanian government for almost 33 years. Originally TASPAWS, by the time he died, it had transmogrified into the Nature Conservation Branch of the Tasmanian Department of Primary Industries, Water and Environment. He used his position genuinely and tenaciously to strive to work towards the conservation of the species (pl. 13), in fact it was his life's work. How many of us can say that we have given our lives for a cause?

As his sister, Eva, noted in her eulogy at his funeral, it could be argued that Irynej's professional interest in seabirds in general and muttonbirds in particular was shaped by four islands. The first is the island of Tasmania, where he was born and grew up and which ignited his love of the natural environment. The second is Macquarie Island, where he conducted his first scientific fieldwork and began his professional career as a scientist. The third is Fisher Island, where his long-term study of the population dynamics of muttonbirds, established him as a scholar of international repute. And the fourth is The Snares, which he believed contained the secrets of the habits of several seabirds (Skira, E. 2005). It is a tragic irony that he was struck down on the island that he had longed to visit.

A quiet, gentle, softly spoken, unassuming and unique man, with a dry, ironic sense of humour, Irynej's devotion to environmental science was greatly admired and respected by his peers. Henrik Moller noted his "gentle, tolerant, generous nature — a real gem of a man as well as a fine scholar" (Moller 2005: 4). Nick Mooney believed that his "extra effort born of enthusiasm, curiosity, discipline and a positive attitude", "those extra yards and minutes", got Irynej his achievements (Mooney 2005: 5). His PhD supervisor, Peter Hay, told me he had the attributes of a Renaissance scholar. He was a first-class scientist, widely read across a huge range of disciplines, with a deep love of music, painting and photography and open to new ways of studying the world. His wife Suzanne noted that "he moved gently on the earth but left behind a huge legacy". He showed how one person's commitment and personal endeavour can make a difference. It is a tragedy that at the moment when he found personal happiness and achieved international recognition, his life was cut short. His premature death brought to an end a remarkable and passionate relationship with nature.

The chief legacy of Irynej's work was to demonstrate the immense importance of long-term monitoring of seabird populations (pl. 14). A quote from the paper Suzanne presented to the conference in Belfast in August 2005 forcefully shows this:

*PLATE 13*
*Irynej Skira with a young Short-tailed Shearwater, the species whose conservation formed a large part of his scientific life.*

*PLATE 14*
*Irynej Skira with a Short-tailed Shearwater removed from its burrow as a part of the long-term monitoring programme.*

*Short-tailed Shearwater populations are difficult to gauge without having access to long-term data sets. More than 50 years of data are available on Short-tailed Shearwaters on Fisher Island, but the small breeding population makes it difficult to determine whether large-scale climate influences or others factors are influencing the population. Nevertheless, such data serve as a starting point. Seabirds are an important indicator of changes in global climate. Long-term data sets are the crucial answer to the problems of seabird population fluctuations.*

But, long-term monitoring needs long-term support and this can probably only come from government. But in Irynej's view anyone contemplating long-term study should also actively seek publicity in order to anchor their research in the public mind. It then makes it harder to terminate. The publicity generated by shearwaters, largely through their own amazing lifestyle (pl. 15), has always been of great media interest and has helped keep the work alive (Skira 2005b: 6).

This issue is homage to this remarkable man and to his extraordinary work.

*PLATE 15*
*Short-tailed Shearwaters,* Puffinus tenuirostris, *leaving their breedings grounds in Tasmania in April for their trans-equatorial migration.*

## ACKNOWLEDGEMENTS

I would like to thank Peter Hay and Clive Tilsely for sharing their memories of Irynej and commenting on earlier drafts. I would particularly like to thank Eva Skira for sharing her memories of Irynej's childhood, providing a copy of her funeral oration on Irynej and for her sound editorial skills. I would also like to thank Nick Mooney for allowing me to use his eulogy and obituary of Irynej, to flesh out his scientific career. Above all I would like to thank Suzanne Skira for sharing her memories of Irynej's later life and sending me copies of his publications as well as her own and others, to understand his scientific legacy. Any mistakes or oversights are my own.

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# VERTEBRATE FAUNA ASSOCIATES OF THE WEDGE-TAILED SHEARWATER, *PUFFINUS PACIFICUS*, COLONIES OF ROTTNEST ISLAND: INFLUENCE OF AN ECOSYSTEM ENGINEER

by Wesley J. Bancroft, J. Dale Roberts and Mark J. Garkaklis

(with five text-figures and three tables)

**Bancroft, W.J., Roberts, J.D. & Garkalis, M.J.** 2008 (31:x): Vertebrate fauna associates of the Wedge-tailed Shearwater, *Puffinus pacificus*, colonies of Rottnest Island: influence of an ecosystem engineer. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 21–30. ISSN 0080-4703. School of Animal Biology, M092, University of Western Australia, Crawley WA 6009, Australia (WJB, JDR, MJG), and 3 Gregona Place, Kalamunda WA 6076, Australia (WJB*). *Author and address for correspondence. Email: wes@graduate.uwa.edu.au.

Wedge-tailed Shearwaters, *Puffinus pacificus*, engineer the ecosystem by digging burrows in which they nest. This has been previously shown to affect the soil and vegetation properties of their colonies. Here we report on field surveys employed to investigate how associated vertebrate fauna respond to physical habitat modification by shearwaters. The study area was species poor, with only one mammal, and three reptile species detected in 1440 Elliott trap and 720 pitfall trap nights across a 13-month period. Nineteen bird species were recorded from 98 survey days. Relative to an area of uncolonised heath, we observed an increase in the abundance of King's Skinks, *Egernia kingii*, and a decrease in the abundance of House Mice, *Mus musculus*, and West Coast Ctenotus, *Ctenotus fallens*, in the shearwater colony. The survival rates of King's Skinks and House Mice were not affected by Wedge-tailed Shearwater presence. Bird species richness was less in the colony (9.2 ±0.5 species month$^{-1}$) than the heath (11.5 ±0.2 species month$^{-1}$), and the composition of the two communities was different. We suggest that ecosystem engineering by Wedge-tailed Shearwaters is a major determinant of fauna associates of their colonies and offer direct and indirect mechanisms to explain the patterns of species occurrence observed.

**Key Words: Wedge-tailed Shearwater, *Puffinus pacificus*, ecosystem engineering, burrow, Rottnest Island, avifauna, *Egernia kingii*, *Mus musculus*, *Ctenotus fallens*.**

## INTRODUCTION

Wedge-tailed Shearwaters, *Puffinus pacificus* (J.F. Gmelin, 1789), are pelagic seabirds that nest in burrows on islands in the Indian and Pacific oceans. They have been described as ecosystem engineers because they disrupt resource flows within their colonies as a consequence of their physical modification of habitat (Jones *et al.* 1994, Bancroft *et al.* 2004b). On Rottnest Island, Western Australia, burrowing by an expanding colony of Wedge-tailed Shearwaters has created 5.1 km of tunnels ha$^{-1}$, displaced 210 t soil ha$^{-1}$ and significantly modified physical and chemical soil properties (Bancroft *et al.* 2004a, b, Bancroft *et al.* 2005a, b). These soil perturbations correlate with changes in plant diversity, structure and productivity (Bancroft *et al.* 2005c). The impact that burrowing by the birds has on the island fauna has yet to be investigated.

Burrowing may affect fauna by two main mechanisms: directly, by providing a habitat resource, the burrow; and indirectly, by altering the edaphic or botanical characteristics of a site via biopedturbation (Jones *et al.* 1994, 1997, Kinlaw 1999, Whitford & Kay 1999). These mechanisms have been best studied in mammals and the direct use of mammalian burrows by non-burrowing taxa has been well documented (e.g., mole, *Scalopus* spp.; prairie dog, *Cynomys* spp.; rat, *Eliurus* spp.; and kangaroo rat, *Dipodomys* spp. systems; Scheffer 1945, Campbell & Clark 1981, Hawkins & Nicoletto 1992, Goodman 1994). Burrows can benefit associated animals by providing shelter from predators or climate, a site for socialisation or communication, or by conveying physiological advantages (Kinlaw 1999). Avian burrows are also directly utilised by co-occurring fauna: Tiger Snakes, *Notechis scutatus* (Peters, 1861), cohabit Short-tailed Shearwater, *Puffinus tenuirostris* (Temminck, 1835), burrows (Serventy *et al.* 1971); Tuatara, *Sphenodon punctatus* (Gray, 1842), utilise Fairy Prion, *Pachyptila turtur* (Kuhl, 1820), burrows (Newman 1987); and King's Skinks, *Egernia kingii* (J.E. Gray, 1838), and Dibbler, *Parantechinus apicalis* (J.E. Gray, 1842), use Wedge-tailed Shearwater burrows (K. Wolfe, pers. comm., Chapple 2003). The indirect impacts of burrowing mammals on faunal communities are predominantly through the disruption of food webs, and have been associated with changes in species diversity (Whicker & Detling 1988, Ceballos *et al.* 1999, Kinlaw 1999) and abundance (Whicker & Detling 1988, Hawkins & Nicoletto 1992). Nutrient and trophic cascades have been implicated in the increased abundance of fauna in burrowing seabird colonies (Markwell & Daugherty 2002) and post-breeding survival of male Dibbler (Wolfe *et al.* 2004) but otherwise the impact of avian ecosystem engineers is poorly known.

Here we follow our previous studies of the ecosystem engineering impact of Wedge-tailed Shearwaters on soil and vegetation by determining the vertebrate fauna associates of their colonies on Rottnest Island. We used Elliott and pitfall trapping to survey for amphibians, reptiles and mammals, and area searches for avifauna. Capture-mark-recapture techniques were used to estimate population size and survival rates for the two dominant, non-avian vertebrates: House Mice, *Mus musculus* Linnaeus, 1758, and King's Skinks. Areas with and without shearwater burrows were compared to investigate the impacts that ecosystem engineering by shearwaters has on the abundance and species composition of the associated fauna community.

**TABLE 1**
**The percentage of surveys in which bird species were recorded present on, in or above the colony and heath**

| | Colony | | | | | | | | | | | | Heath | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Mar 2003 | Apr 2002 | May 2002 | Jun 2002 | Jul 2002 | Sep 2002 | Oct 2002 | Dec 2002 | Feb 2003 | Mar 2003 | Apr 2003 | Overall | Mar 2003 | Apr 2002 | May 2002 | Jun 2002 | Jul 2002 | Sep 2002 | Oct 2002 | Dec 2002 | Feb 2003 | Mar 2003 | Apr 2003 | Overall |
| *Terrestrial species* | | | | | | | | | | | | | | | | | | | | | | | | |
| Ring-necked Pheasant, *Phasianus colchicus* | – | – | – | – | – | – | – | – | – | – | – | – | 91 | 80 | 100 | 100 | 89 | 100 | 100 | 90 | 100 | 75 | 100 | 93 |
| Wedge-tailed Shearwater, *Puffinus pacificus* | 100 | 100 | 25 | | | 75 | 78 | 100 | 100 | 100 | 100 | 71 | – | – | – | – | – | – | – | – | – | – | – | – |
| Mountain Duck, *Tadorna tadornoides* | – | – | – | 50 | 67 | 25 | – | – | – | – | 20 | 15 | – | – | – | – | – | – | – | – | – | – | – | – |
| Painted Button-quail, *Turnix varia* | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | 22 | – | 11 | – | 22 | 17 | – | – | 7 |
| Ruddy Turnstone, *Arenaria interpres* | 100 | – | – | – | – | – | – | – | – | – | 100 | 18 | – | – | – | – | – | – | – | – | – | – | – | – |
| Pied Oystercatcher, *Haematopus longirostris* | 9 | – | 43 | 25 | 56 | 38 | – | – | – | – | 100 | 25 | – | – | – | – | – | – | – | – | – | – | – | – |
| Silver Gull, *Larus novaehollandiae* | – | – | 14 | 75 | 100 | 75 | 33 | 10 | – | – | 80 | 35 | – | – | – | – | – | – | 11 | – | – | – | – | 1 |
| Laughing Turtle-dove, *Streptopelia senegalensis* | – | – | – | – | – | – | – | – | – | – | – | – | 55 | 100 | 86 | 75 | 89 | 88 | 100 | 100 | 89 | 100 | 100 | 89 |
| White-browed Scrubwren, *Sericornis frontalis* | – | – | – | 25 | – | – | – | – | 11 | – | – | 4 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| Singing Honeyeater, *Lichenostomus virescens* | 18 | 20 | 71 | 100 | 33 | 25 | – | 40 | – | – | 80 | 35 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| White-fronted Chat, *Epthianura albifrons* | 100 | 100 | 100 | 100 | 33 | 75 | 89 | 90 | 44 | 25 | 100 | 78 | | | 29 | 75 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 73 |
| Australian Raven, *Corvus coronoides* | 55 | 60 | 71 | – | 11 | 13 | 33 | 10 | – | – | – | 23 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 80 | 100 | 75 | 100 | 96 |
| Australasian Pipit, *Anthus novaeseelandiae* | – | – | – | – | – | – | – | – | – | – | – | – | 18 | – | – | – | – | – | – | – | – | – | 20 | 3 |
| Silvereye, *Zosterops lateralis* | – | – | – | 50 | – | – | 11 | 30 | – | – | 40 | 12 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| *Aerial species – foraging* | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Osprey, *Pandion haliaetus* | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| Black-shouldered Kite, *Elanus axillaris* | – | – | – | – | – | – | – | – | – | – | – | – | 27 | – | – | – | – | – | – | – | – | – | – | 5 |
| Nankeen Kestrel, *Falco cenchroides* | 18 | 13 | – | 25 | – | 13 | 33 | – | 11 | 33 | – | 13 | 27 | 25 | 10 | 38 | – | 25 | 56 | 20 | 22 | 33 | 30 | 26 |
| Welcome Swallow, *Hirundo neoxena* | 100 | 100 | 100 | 100 | 78 | 88 | 89 | 80 | 100 | 100 | 80 | 91 | 100 | 100 | 100 | 88 | 78 | 100 | 89 | 80 | 100 | 100 | 80 | 92 |
| Tree Martin, *Hirundo nigricans* | 91 | 100 | 90 | 100 | 78 | 100 | 100 | 70 | 100 | 83 | 90 | 92 | 91 | 100 | 90 | 100 | 89 | 88 | 100 | 80 | 100 | 83 | 90 | 92 |
| *Aerial species – in transit* | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Crested Tern, *Sterna bergii* | 45 | 50 | 30 | 13 | 11 | 75 | 67 | 60 | 67 | 50 | 40 | 46 | 45 | 38 | 30 | 25 | 11 | 75 | 56 | 50 | 56 | 33 | 40 | 42 |
| Fairy Tern, *Sterna nereis* | – | 13 | – | – | – | – | 33 | 50 | – | – | – | 9 | – | – | – | – | – | – | 33 | 50 | – | – | – | 8 |
| Fork-tailed Swift, *Apus pacificus* | 9 | – | – | – | – | – | – | – | – | – | – | 1 | 9 | – | – | – | – | – | – | – | – | – | – | 1 |

## MATERIALS AND METHODS

The study was conducted on Rottnest Island (32°00'S, 115°31'E), southwest Western Australia (fig. 1), between March 2002 and April 2003, and focused on the largest two Wedge-tailed Shearwater colonies on the island, Radar Reef, and Cape Vlamingh (Bancroft *et al.* 2004a). Colonies were vegetated by low, ruderal and, predominantly, non-native species (Bancroft *et al.* 2005c). Plant species diversity on the colonies was low, with much of the colony surface dominated by a monoculture mat of Iceplant, *Mesembryanthemum crystallinum* L. (Bancroft *et al.* 2005c). Surrounding uncolonised areas comprised low (c. 1 m), moderately diverse, native heath vegetation (Bancroft *et al.* 2005c). Detailed descriptions of the vegetation of the colony and non-colony (heath) areas are provided by Bancroft *et al.* 2005c.

A single trapping location was selected (using randomly generated mapping co-ordinates) in each colony, and in the heath within c. 50 m of each colony's boundary. Birds were surveyed by area searches and non-avian, terrestrial vertebrates were surveyed by both pitfall and Elliott trapping. The pitfall trapping was limited to the Radar Reef colony because there was a risk of public interference with the pitfall traps at Cape Vlamingh. Both trapping methods exclude the capture of larger animals (see trap dimensions below). The only such species that was known to occur within the study area was the Quokka, *Setonix brachyurus* (Quoy & Gaimard, 1830). A pilot study of Quokkas suggested that their abundance was too low to warrant further investigation.

### Bird surveys

Bird species were recorded as present/absent in the colony and heath while conducting Elliott and pitfall trapping, and other work. The area of heath surveyed was equivalent to the respective colony area (approximately 4 ha). Birds were surveyed on 98 days over a period of 13 months: March (11 days), April (8 days), May (10 days), June (8 days), July (9 days), September (8 days), October (9 days), and December 2002 (10 days), and February (9 days), March (6 days) and April 2003 (10 days). Recordings were made each day between 0700 and 0900, and 1400 and 1600. Species were classified as: (i) terrestrial, if observed predominantly on the heath or colony locations, (ii) aerial – foraging, if observed on the wing above the locations and were using them as a foraging resource, or (iii) aerial – in transit, if predominantly observed above the locations in transit. The three aerial – in transit species (see table 1) were excluded from subsequent analyses.

Occurrence data were used to calculate Shannon-Weiner diversity index ($H$), and the equitability ($E$) for the colony and heath communities (see Krebs 1999). Similarly, two community comparison indices were calculated; the coefficient of Jaccard (Sj) and percentage similarity ($P$, see Krebs 1989). Data from each month were treated separately.

### Elliott trapping

At each trapping location, 12 Elliott traps (325 x 85 x 95 mm treadle, box-traps; Elliott Scientific, Upwey Victoria) were arranged in a line at 5 m intervals. Traps were baited with a mix of rolled oats, sardines and peanut butter. Traps were opened for five nights on six survey occasions: March, May, July, October, December 2002 and February 2003. Traps were checked in the early morning. Captured animals were individually marked by toe- (reptiles) or ear-clipping (mammals), and immediately released.

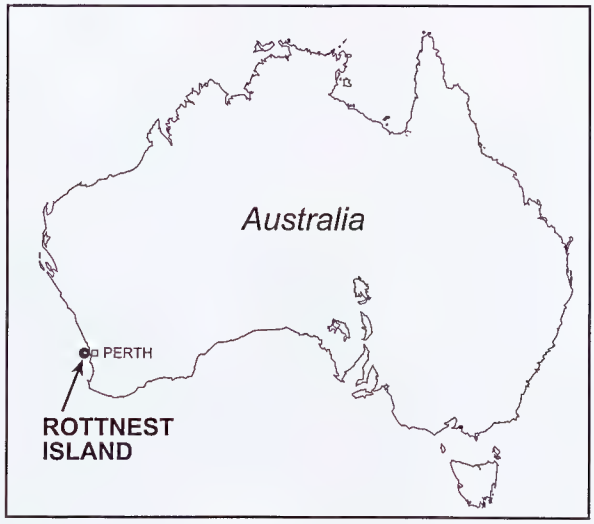


*FIG. 1 — Location of Rottnest Island.*

Data from both trapping locations within the colony or heath were pooled for analysis. Chi-squared ($\chi^2$) tests were used to analyse frequency data. Repeated-measures (RM) ANOVA was used to analyse capture rates. MANOVA was used to concurrently analyse the new capture and recapture rates of multiple species (variables). The Jolly-Seber model equations presented by Krebs (1999) and Program MARK (Gary White, Colorado State University) were used to derive population estimates, and survival and recapture probabilities from the capture-mark-recapture data. Assumptions of this model are provided by Krebs (1999). Student t-tests were used to compare population estimates (paired in time) and movements of colony and heath animals.

### Pitfall trapping

At each location, 12 unfenced, unbaited, PVC pitfall traps (150 mm diameter x 400 mm depth) were spaced in a line at 5 m intervals. As the traps were initially dug in, wire mesh was placed beneath the trap to prevent animals from burrowing out of the traps, while allowing free drainage of water. Traps were opened for four consecutive nights in each of June, July, October, December 2002 and February 2003. Traps were checked in the early morning. When not in use, traps were sealed with metal lids and covered with soil. Captured animals were released, unmarked.

Reptile data (all species) were pooled for analyses. Total reptile and mouse abundances for each location were compared using a $\chi^2$ test. RM-ANOVA was used to analyse capture rates.

### Burrow use by King's Skinks

Internally-winding cotton spools were attached to the base of the tail of 14 King's Skinks (six at Cape Vlamingh, eight at Radar Reef) trapped by Elliott trapping in December 2002 (during the period of highest reptile activity). The free end of the cotton was tied to vegetation adjacent to the point of capture and the animal released. We returned 24 hours later, recaptured the animal, and used the cotton tracer to determine the number of burrows it had entered.

## RESULTS

### Bird surveys

Twenty-two species were recorded on, in or over the colony or heath. Of these, 19 were observed using the colony or heath as a resource (terrestrial and aerial-foraging species; table 1). Ten species were recorded on both the colony and heath, with four exclusively in the colony: Wedge-tailed Shearwater; Mountain Duck, *Tadorna tadornoides* (Jardine & Selby, 1828); Ruddy Turnstone, *Arenaria interpres* (Linnaeus, 1758); and Pied Oystercatcher, *Haematopus longirostris* Vieillot, 1817. Five species were recorded exclusively in the heath: Ring-necked Pheasant, *Phasianus colchicus* Linnaeus, 1758; Painted Button-quail, *Turnix varia* (Latham, 1802); Laughing Turtle-dove, *Streptopelia senegalensis* (Linnaeus, 1766); Australasian Pipit, *Anthus novaeseelandiae* Vieillot, 1818; and Black-shouldered Kite, *Elanus axillaris* (Latham, 1802). By definition of our locations, Wedge-tailed Shearwaters were only observed in the colony and our records reflect birds present in burrows during the survey period. Mountain Ducks were mainly observed in winter months, when up to six birds foraged through the low colony vegetation. Ruddy Turnstones were present on the colony in both the autumn of 2002 and 2003, but at all other times of the year were absent from the location. Flocks of up to 40 turnstones foraged over the colony surface, frequently probing beneath senescent vegetation for invertebrates. Similarly, Pied Oystercatchers foraged, usually in pairs or threes, in the colony during autumn, but also extended their use of the location through winter.

Based on species richness, the two communities were moderately similar (co-efficient of Jaccard; $S_j$ = 0.53). Accounting for the frequency of species presence, the communities were still similar (percentage similarity, $P$ = 50.8%). Both locations were similar in diversity (Shannon-Weiner diversity index, $H$); for the colony $H$ = 0.91 and for the heath $H$ = 0.96. Species were evenly encountered (equitability index, $E$) at each location; for the colony $E$ = 0.79 and for the heath, $E$ = 0.81. The mean monthly species richness was significantly less in the colony (9.2 ±0.5, range 6–11) than the heath (11.5 ±0.2, range 10–13; paired t-test, $t_{10}$ = 4.1, $P$ = 0.002).

### Elliott trapping

Three measures were used to assess the relative use of colony and heath locations by animals captured by Elliott trapping: total individuals captured, capture rate and estimated population size. Three species were captured in the 1440 Elliott trap nights: King's Skink (91 individuals, 501 captures), House Mouse (287 individuals, 469 captures) and West Coast Ctenotus, *Ctenotus fallens* Storr, 1974, (five individuals, five captures). West Coast Ctenotus was excluded from analyses due to the low number of captures.

Totalled across all surveys, there were significantly more King's Skinks (52) and fewer House Mice (115) individuals in the colony than in the heath (39 and 172 respectively; $\chi^2$ = 8.2, d.f. = 1, $P$ = 0.004). The mean number of animals (new and recaptured) captured trap$^{-1}$ night$^{-1}$ showed the same, significant trends for both King's Skink (RM-ANOVA, $F_{1,94}$ = 8.3, $P$ <0.001) and House Mouse (RM-ANOVA, $F_{1,238}$ = 11.8, $P$ = 0.001; fig. 2). Subsequent analysis treated new and recaptured animals as separate variables and showed that this trend was attributable to differences in rates of capture of both groups (MANOVA, $F_{4,1425}$ = 10.72, $P$ <0.001). There was also a significant effect of trapping month on capture rate (MANOVA, $F_{10,2854}$ = 8.96, $P$ <0.001; fig. 2). Capture rate differed significantly between all months, with the exception of October and December 2002 for King's Skink (SNK, $P$ = 0.767), and March 2002 and February 2003 (SNK, $P$ = 0.817) for House Mouse. There was a peak in the capture rate of House Mice in the winter (July) survey, and the maximum number of King's Skinks was caught in spring and early summer (October and December). Colony and heath capture rates behaved consistently over time (no significant interaction between location and month; MANOVA, $F_{10,2854}$ = 0.84, $P$ = 0.587). While overall there was a significantly higher capture rate of House Mice in the heath, during July the capture rate in the colony rose to equal that of the heath (SNK, $P$ = 0.294; fig. 2).

Analysis of capture-mark-recapture data produced four separate estimates of population size for both species at each location (fig. 3). The colony supported a larger population of King's Skinks (paired t-test, $t_3$ = 11.0, $P$ = 0.002), and a smaller population of House Mice ( $t_3$ = 3.4, $P$ = 0.044) than the heath. Comparison of the means and standard errors presented in table 2 suggests that, within species, there was no difference in the survival and recapture probabilities of colony and heath animals.

Seventy King's Skinks and 118 House Mice were recaptured on at least one occasion. Frequency distributions of the maximum distance moved by recaptured animals are shown in figure 4. For individuals that moved only within a trapping location (see fig. 4), the mean distance moved did not differ for colony and heath animals (King's Skink, $t_{59}$ = 0.4, $P$ = 0.680; House Mouse, $t_{105}$ = 0.4, $P$ = 0.700). For these animals, the average maximum distance moved (direct distance between traps) was 11.6 ±1.1SE m for King's Skink (range 0–32.4, n = 61) and 15.0 ±1.3SE m for House Mouse (range 0–55.4, n = 107). Nine King's Skinks (9.9% of all individuals captured) and 11 House Mice (3.8%) moved between trapping locations. Two movements of King's Skinks were of particular note. One animal moved at least 770.7 m (between July and October 2002), and the other 730.5 m (between December 2002 and February 2003). The remaining King's Skink moved between 56.3 and 76.9 m (fig. 4). Between-location movements by House Mice ranged from 55.3 to 81.1 m. Animals originally marked in the colony and heath did not differ in their between-location movements (t-test, King's Skink, $t_7$ = 0.1, $P$ = 0.922; House Mouse, $t_9$ = 1.8, $P$ = 0.110).

### Pitfall trapping

One species of mammal (House Mouse) and three species of reptile (West Coast Ctenotus; King's Skink; and Western Pale-flecked Morethia, *Morethia lineoocellata* (Duméril & Bibron, 1839)) were captured in the 720 trap nights (numbers of captures are presented in table 3). Species richness was higher in the heath (3) than the colony (2). The total number of captured House Mice and reptiles (pooled captures) differed significantly between the three locations surveyed by pitfall trapping ($\chi^2$ = 23.9, d.f. = 1, $P$ < 0.001). There were more mice in the colony (45 captures) than the heath (22), and more reptiles in the heath (19) than the colony (1).

The rate of capture (animals trap$^{-1}$ night$^{-1}$) differed significantly between locations for House Mice (RM-ANOVA, $F_{1,94}$ = 5.1, $P$ = 0.027), and reptiles (RM-ANOVA, $F_{1,94}$ = 14.1, $P$ < 0.001; fig. 5). For both groups there was a

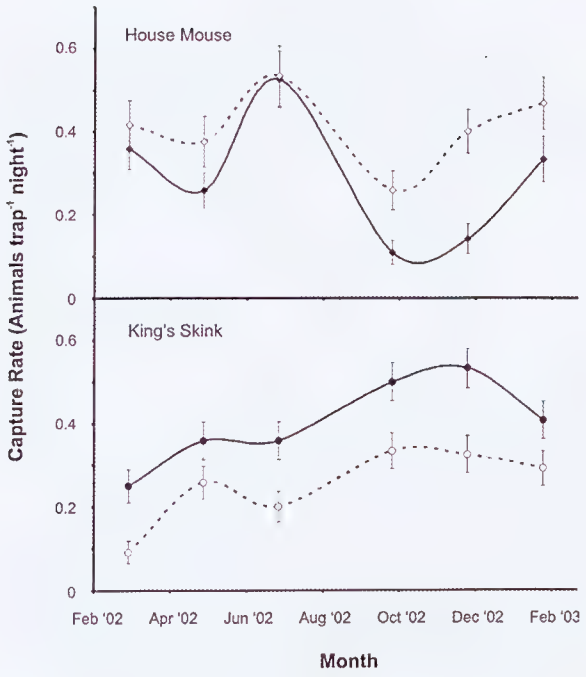


*FIG. 2 — The rate of capture of King's Skinks and House Mice in the colony (solid line) and heath (dashed line) for the six Elliott trapping surveys. Error bars represent standard errors.*

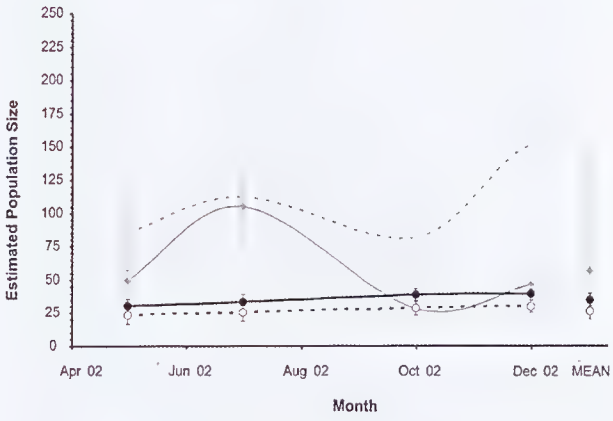


*FIG. 3 — Estimated total population sizes of King's Skinks (circles) and House Mice (diamonds) in the colony (solid points and lines) and heath (open points and dashed lines) areas surveyed. Error bars represent standard errors.*

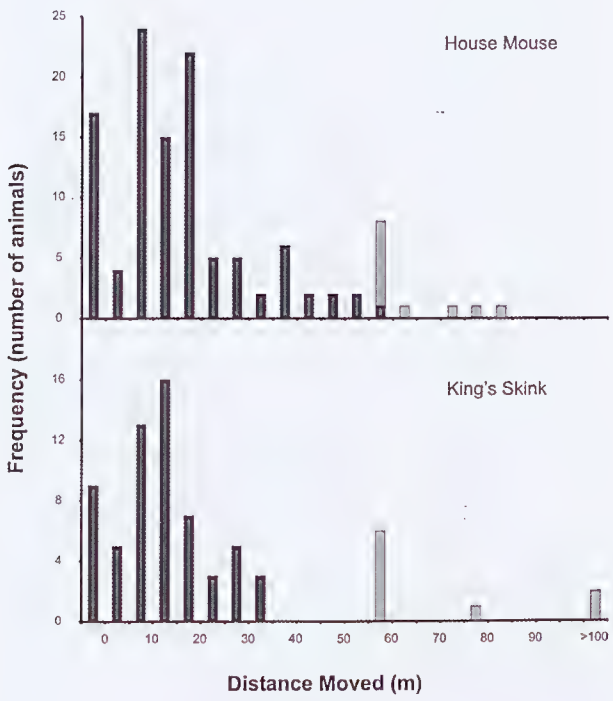


*FIG. 4 — Frequency histogram of the maximum distance moved within (black bars) and between (grey bars) trapping locations by King's Skinks (N =70) and House Mice (N =118).*

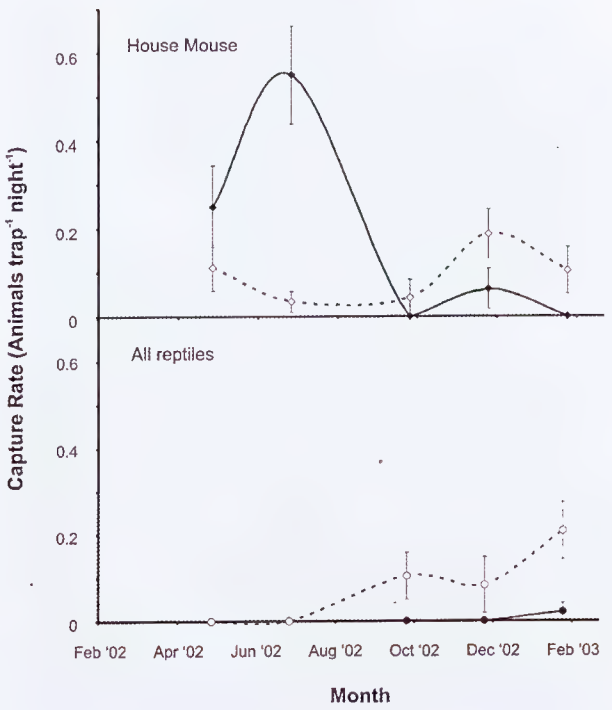


*FIG. 5 — The rate of capture of House Mice and all reptiles in the colony (solid line), and heath (short-dashed line) for the five pitfall trapping surveys. Error bars represent standard errors.*

**TABLE 2**
**Annual probability of survival and recapture for King's Skink and House Mouse in the colony and heath**

| Species | Location | Probability of survival | Probability of recapture |
|---|---|---|---|
| King's Skink, *Egernia kingii* | Colony | 0.906 ±0.032 (0.823–0.952) | 0.744 ±0.045 (0.645–0.822) |
| | Heath | 0.930 ±0.037 (0.814–0.978) | 0.676 ±0.059 (0.551–0.780) |
| House Mouse, *Mus musculus* | Colony | 0.676 ±0.082 (0.501–0.813) | 0.234 ±0.061 (0.135–0.374) |
| | Heath | 0.676 ±0.055 (0.560–0.774) | 0.324 ±0.054 (0.228–0.438) |

Values given ±SE. 95% confidence limits in parentheses

significant effect of trapping occasion (RM-ANOVA, $F_{4,94} > 7.1$, $P < 0.001$ for both cases).

### Burrow use by King's Skinks

On average King's Skinks utilised 6.07 ±0.27SE burrows lizard$^{-1}$ day$^{-1}$ (range 4–8).

## DISCUSSION

Wedge-tailed Shearwaters on Rottnest Island have engineered the vegetation from a reasonably diverse, waist-height heath to a low mat of vegetation that is dominated by Iceplant (Bancroft *et al.* 2005c). It is highly likely that these changes in vegetation structure and composition are responsible for the differences in avifauna species richness and composition that we observed. Two shorebird species that are usually associated with the beaches and salt lakes were attracted to the colonies. Senescent colony vegetation produced conditions that were similar to the rotting seaweed banks that are often frequented by these species (Saunders & de Rebeira 1993). While the low, open areas of shearwater colony attracted a different guild of foraging birds from the adjacent heath, it also excluded other modes of foraging. Ground feeding granivores such as Ring-necked Pheasant, Laughing Turtle-dove and Painted Button-quail were not present in the colony. Iceplant produces very small, wind-blown seeds that are unlikely to support these birds, whereas heath provides a greater diversity of larger seed-bearing species (Bancroft *et al.* 2005c). The granivores, especially Ring-necked Pheasant and Painted Button-quail, prefer cryptic, enclosed habitat (Marchant & Higgins 1993, Saunders & de Rebeira 1993) so would be expected to avoid the open, exposed colony.

King's Skinks and House Mice dominated the trapping studies. The two species appear to respond to the presence and physical impact of Wedge-tailed Shearwaters (Bancroft *et al.* 2004b, Bancroft *et al.* 2005a) in opposing manners. The shearwater colonies supported a greater number of King's Skinks (as measured by number of new animals, individuals, capture rate and estimated population size) than the adjacent heath, but there were fewer House Mice in the colony than the heath. Given the massive modification of habitat, and the disruption of edaphic and vegetation characteristics of the colonies via biopedturbation and burrowing (Bancroft *et al.* 2004a, b, 2005a, b, c), it is highly likely that ecosystem engineering by the shearwaters is responsible (at least in part) for these patterns. Several possible mechanisms, direct and indirect, may explain the patterns observed.

**TABLE 3**
**The total number of captures of species caught by pitfall trapping at each of the two locations**

| Species | Locations | |
|---|---|---|
| | Colony | Heath |
| House Mouse, *Mus musculus* | 45 | 22 |
| West Coast Ctenotus, *Ctenotus fallens* | – | 17 |
| King's Skink, *Egernia kingii* | – | 2 |
| Western Pale-flecked Morethia, *Morethia lineoocellata* | 1 | – |

The two major, direct functions of burrows are to provide shelter from the environment or predators, and to convey physiological benefits (Reichman & Smith 1990, Kinlaw 1999). King's Skink is an ectothermic reptile; it behaviourally thermoregulates and is therefore dependent on the surrounding environment for sources of heat or shade (Withers 1992). The shearwater colonies would be particularly favourable locations for this species because the burrows provide a relatively constant temperature environment and an excellent source of shade in hot conditions (Whittow *et al.* 1987). Throughout the year Rottnest Island burrow temperatures range from 13.2° to 22.7°C (W. Bancroft, unpubl. data), but the ground surface temperatures range from 10.8° to 47.3°C (Bancroft *et al.* 2004b, 2005a). It is known that other reptiles, such as snakes, use burrows for thermoregulation (Serventy *et al.* 1971, Whitaker & Shine 2002) and we frequently observed King's Skinks shuttling in and out of burrows. The House Mouse is an endothermic mammal and is less dependent on the surrounding environment to thermoregulate. While burrows can certainly assist the thermal physiology of endothermic animals they are by no means essential for House Mice (Withers 1992, Moro & Morris 2000). Both species may also use burrows to avoid predation (Strahan 1995, Cogger 2000). On Rottnest Island, the major predators of both species are likely to be birds of prey (Saunders & de Rebeira 1993) and burrows are particularly effective for avoiding aerial predators (Kinlaw 1999). While House Mice readily construct their own burrow systems (Avenant & Smith 2003) and numerous mouse burrows were observed throughout the study, King's Skinks prefer the burrows of other animals (Triggs 1996, Chapple 2003). Again, House Mice are somewhat independent of the surrounding physical habitat resources.

Biopedturbation by shearwaters may indirectly affect the cohabiting fauna via changes to the soil and flora, and flow-on effects through food webs. King's Skink is omnivorous and feeds on a wide variety of foods including vegetation, invertebrates, carrion and seabird eggs (Bush *et al.* 1995, Chapple 2003). The House Mouse is also omnivorous but on islands its diet is more limited than King's Skink and

comprises mostly invertebrates, with some supplementary plant material (Le Roux *et al.* 2002, Moro & Bradshaw 2002, Smith *et al.* 2003). Shearwaters reduce plant species diversity, and the structural complexity of colony vegetation (Bancroft *et al.* 2005c) and this may reduce arthropod diversity (Abbott 1976). Such a mechanism may indirectly limit the House Mouse population and the broader dietary range of King's Skinks may buffer this species against perturbations in food supply caused by shearwater biopedturbation. Wolfe (2004), however, noted an increased abundance of invertebrates in the shearwater colonies on Whitlock and Boullanger islands (c. 200 km to the north of Rottnest Island). Markwell & Daugherty (2002) also found an increase in invertebrate abundance on New Zealand islands inhabited by burrowing seabirds. If this is true for Rottnest Island, then other processes must be acting to limit mouse numbers. Regrettably we have no data on invertebrate abundance or diversity in the Rottnest Island shearwater colonies to test this, but the seasonal pattern in House Mouse abundance supports its exclusion by shearwaters. The House Mouse population in the colony was suppressed (relative to the heath) for the majority of the year, except for the winter survey, where the population rose to equal that of the heath (see figs 2, 3). This is also the period when the shearwaters were completely absent from the island. Absence of the shearwaters apparently permits the colony House Mouse population to reach the level recorded in the heath.

Indeed, the direct and indirect effects of the burrows or biopedturbation may be overlain with direct interactions between the burrower and the cohabiting fauna. King's Skinks are diurnal and House Mice are predominantly nocturnal (Strahan 1995, Chapple 2003). The majority of shearwaters forage at sea during the day and return to their colony after dark (Marchant & Higgins 1990) so there would be relatively little interaction with active King's Skinks but considerable overlap with active House Mice. Predation of mice or skinks by Wedge-tailed Shearwaters has not been reported, but House Mice fall within the size range of fish taken by the birds, so may be a potential food source (Barker & Vestjens 1989, Strahan 1995, Schultz & Klomp 2000). The Sheathbill, *Chionis minor* Hartlaub, 1841, is of a similar body size to the Wedge-tailed Shearwater and is known to predate on mice (V.R. Smith, pers. comm.). At c. 220 g, adult King's Skinks are unlikely to be consumed by shearwaters (c. 380 g, Dunning 1993, Storr *et al.* 1999, Schultz & Klomp 2000). Both skinks and mice feed on seabird eggs or attack chicks (Bush *et al.* 1995, Cuthbert & Hilton 2004) so may be a potential threat to nesting shearwaters. Short-tailed Shearwaters appear to tolerate predators (Tiger Snakes, Serventy *et al.* 1971) although there are several reports of other burrowers defending their burrows and evicting snakes (Halpin 1983, Newman 1987, Randall *et al.* 2000). Wedge-tailed Shearwaters apparently tolerated King's Skinks, as many of the burrows utilised by the skinks were also actively used by the birds, including incubating adults.

Our data, therefore, show that Wedge-tailed Shearwaters had a significant effect on the King's Skink and House Mouse populations on the western end of Rottnest Island but that the main causes of these impacts were most likely different mechanisms. We suggest that the increased King's Skink abundance in the shearwater colonies was most likely a result of physical changes in the environment caused by Wedge-tailed Shearwater biopedturbation (ecosystem engineering; Jones *et al.* 1994, 1997). In contrast, the House Mouse populations within the colonies appeared to be suppressed by the presence of the shearwaters themselves. Some effects of ecosystem engineering by Wedge-tailed Shearwaters on the House Mice populations cannot be completely discounted without further investigation.

There was seasonality to the capture rate of both species. The capture rate of King's Skinks peaked in spring, was consistent across both locations, and presumably reflected the period when the animals were most active (see also How & Shine 1999). The capture rate of House Mice was highest in winter, and was probably driven by an increase in population size as a result of increased food availability. Many plants at the study locations fruit or shed their seed at this time of year (e.g., *Acanthocarpus preissii* Lehm., *Olearia axillaris* (DC.) Benth., *Rhagodia baccata* (Labill.) Moq.; Rippey & Rowland 1995) and there is a rapid increase in the germination and growth of new plants (W. Bancroft, unpubl. data). Invertebrate abundance also increases during the wetter months in this region (Recher *et al.* 1996, Majer *et al.* 2003).

The movements of mice and skinks were not influenced by whether they lived within the colony, or in the heath. We found no difference in the survival rates between mice or skinks that lived in the colony, and lived in the heath. So, while the abundance of King's Skinks and House Mice was affected by Wedge-tailed Shearwaters, the survival rates of these species were not. To our knowledge these are the first empirical data to test the effect that an ecosystem engineer has on a life history trait of another species.

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# NEAREST NEIGHBOUR ANALYSIS AND SPATIAL RELATIONSHIPS OF WEDGE-TAILED SHEARWATER, *PUFFINUS PACIFICUS*, (AVES: PROCELLARIIFORMES) BURROW ENTRANCES AT RADAR REEF, ROTTNEST ISLAND, WESTERN AUSTRALIA

by Wesley J. Bancroft

(with one text-figure and one table)

**Bancroft, W.J.** 2008 (31:x): Nearest neighbour analysis and spatial relationships of Wedge-tailed Shearwater, *Puffinus pacificus,* (Aves: Procellariiformes) burrow entrances at Radar Reef, Rottnest Island, Western Australia. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 31–34. ISSN 0080-4703. School of Animal Biology, M092, University of Western Australia, Crawley WA 6009, Australia. Address for correspondence: 3 Gregona Place, Kalamunda, WA 6076, Australia. Email: wes@graduate.uwa.edu.au

The spatial distribution of Wedge-tailed Shearwater burrow entrances on Rottnest Island, Western Australia, was investigated using the single- and two-sector nearest neighbour methods of point pattern analysis. Both analyses yielded similar results. Mean burrow density was 0.32 ±0.02 SE burrows $m^{-2}$, burrow entrances were not distributed at random and entrances tended towards an even distribution. Evenness of distribution was positively correlated with burrow density. Social and structural factors are likely to be important in determining burrow entrance distribution by Wedge-tailed Shearwaters and there appears to be a density-dependent trade-off between social benefit (aggregated burrow entrances at low densities) and colony stability (evenly distributed burrow entrances at high densities).

**Key Words: Shearwater, burrow, nearest neighbour analysis, Procellariiformes, Rottnest Island, burrow density.**

## INTRODUCTION

Burrow-nesting is common among seabirds of the Procellariiformes (Furness 1991). Benefits of burrowing include shelter from environmental extremes or predation, amplification of vocal signals and physiological advantages of a more stable microclimate (Whittow *et al.* 1987, Warham 1990). Burrowing may, however, destabilise the soil and lead to burrow collapse or erosion of colonies which can reduce breeding success through direct mortality or the loss of breeding sites (Furness 1991, Bancroft *et al.* 2005).

The spatial distribution of procellariiform burrows within a colony may influence the risk of burrow collapse or erosion. Given an homogenous environment, and on the basis of structural stability alone, an even distribution of burrows within a colony would be expected (Butler 1995): the further the burrow entrances are from one another, the less likely they are to collapse (through increased communal traffic near the entrance or by a reduction in the soil supporting the burrow roof). However, other factors such as soil strength (Neil & Dyer 1992), ground debris (Hill & Barnes 1989), vegetation type (Floyd & Swanson 1983, Hill & Barnes 1989) and social interactions or relationships (Dyer & Hill 1990) may also influence the location and distribution of burrows within a colony.

A convenient method for examining the spatial distribution of burrows is the nearest neighbour method of point pattern analysis (Taylor 1977). Nearest neighbour analysis compares the observed mean nearest neighbour distance to an expected mean, under the Poisson distribution, to produce a divergence from randomness value, R (Taylor 1977). The divergence from randomness provides a measure of the aggregated (low R), random (moderate R) or even (high R) nature of the spatial distribution of points.

Two studies have used nearest neighbour analysis to investigate the spatial distribution of burrow entrances within Wedge-tailed Shearwater, *Puffinus pacificus* (J.F. Gmelin, 1789), colonies (Dyer & Hill 1990, 1995). These studies identified contrasting patterns in the distribution of burrow entrances. Dyer & Hill (1990) found burrow entrances were moderately aggregated but burrows tended towards an even distribution in the Dyer & Hill (1995) study.

The nearest neighbour analysis is biased towards aggregation when only a single nearest neighbour measurement (single-sector) is taken from each point (Taylor 1977), as was the case in the Dyer & Hill (1990, 1995) studies. Bias can be reduced by averaging the nearest neighbour distances obtained from each of two "hemispheres" surrounding a burrow (a "two-sector" analysis, Taylor 1977).

Dyer & Hill (1990) also found that there was a strong positive correlation between burrow density and evenness of dispersion. Such a correlation could be driven by an attempt to maximise the distance between burrows in order to attain the highest level of structural stability possible (Butler 1995), yet Dyer & Hill (1990, 1995) argued that social interaction was the key factor in determining where birds position their burrows.

Here I investigated the spatial distribution of Wedge-tailed Shearwater burrow entrances in a colony on Rottnest Island, Western Australia, using a two-sector nearest neighbour analysis. Comparison was made to a single-sector analysis. In addition, Bancroft *et al.* (2004) observed that burrow densities were lower on the edge of the Rottnest Island colonies and hypothesised that position within the colony, and in relation to the edge of the island, affected burrow density. Therefore, the present study also investigated the relationships between burrow distribution, burrow density and the distance of burrows from the edge of the colony and the edge of the island.

## METHODS

### Study site

The study was conducted on Rottnest Island (32°00'S, 115°31'E), southwestern Western Australia (fig. 1), on 4–24 September 2002. The study site, Radar Reef, is the largest of six colonies on the island (Bancroft *et al.* 2004).

### Field methods

Quadrat size may influence nearest neighbour analyses (Dyer & Hill 1990). For consistency, and to facilitate comparison, the 10 m x 3 m quadrats identified as appropriate by Dyer & Hill (1990) were used in this study. Quadrats were located adjacent to one another (end on end) along three systematically spaced transects that ran between the north and south boundaries of the colony.

All burrow entrances within a quadrat were marked, counted and the distance (to the nearest 5 cm) to the nearest neighbouring entrance was measured in two arbitrarily determined sectors: to the south (between compass bearings 90° and 270°), and to the north (between 0° and 90°, and 270° and 360°). Burrow entrances outside the quadrats were used as nearest neighbours, where appropriate, to avoid boundary effects (Dyer & Hill 1990). The location of the centre of each quadrat was recorded using a hand-held GPS.

### Data analysis

Analysis of nearest neighbour measurements followed Taylor (1977). The actual mean nearest neighbour distance, $r_{ak}$, is calculated from the measurements taken from k sectors around each of n points in an area A.

The expected mean nearest neighbour distance, $r_{ek} = \sqrt{k}/2\sqrt{(n/A)}$

The divergence from randomness, $R_k = r_{ak} / r_{ek}$

The maximum theoretical divergence from randomness, $R_{k.max} = 2.1491/\sqrt{k}$

The standard error of $r_{ek}$, $SEr_{ek} = 0.26136/\sqrt{(n2/A)}$

The z variate, $z_R = | r_{ek} - r_{ak} | / SEr_{ek}$

Point distribution can be assessed by comparing $R_k$ with $R_{k.max}$, and by using the z variate to test whether $r_{ak}$ and $r_{ek}$ are significantly different.

GIS (ArcView, version 3.0) was used to calculate the minimum distance between each burrow and the edge of the colony and the edge of the island. Correlation coefficients were calculated to examine the relationships between distribution, burrow density and the distance of burrows from the edge of colony and the edge of the island (SPSS v11.0.0).

## RESULTS

A total of 233 burrows in 24 quadrats were surveyed. The mean burrow density across all quadrats was 0.32 ±0.02 SE burrows $m^{-2}$ (range 0.13–0.60 burrows $m^{-2}$ for individual quadrats).

The mean actual two-sector nearest neighbour distance ($r_{a2}$) for all quadrats was 1.42 ±0.03 SE m (range 1.19–2.49 m). The expected mean of randomly distributed burrow entrances ($r_{e2}$) for two sectors and for the mean burrow density observed in this study was 1.24 ±0.03 SE m. The

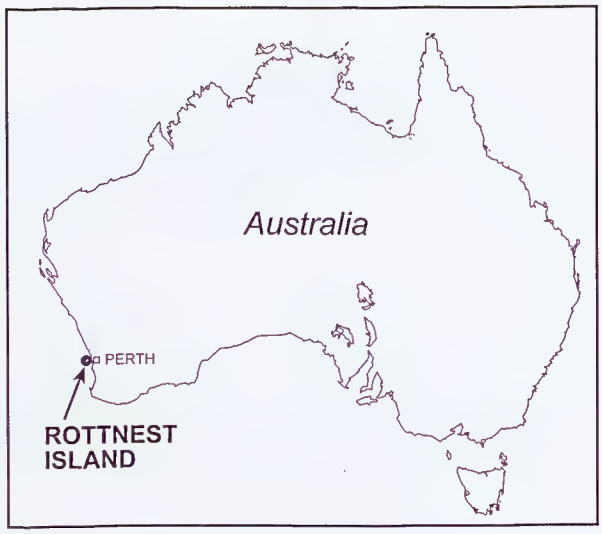


*FIG. 1 — Location of Rottnest Island.*

$r_{a2}$ was significantly greater than the $r_{e2}$ ($z_R = 5.90$, $P < 0.0001$), confirming the non-random distribution and tendency towards maximum possible spacing (evenness).

The maximum theoretical divergence from randomness for two sectors ($R_{2.max}$), where all burrow entrances are evenly dispersed, is 1.52. The observed $R_2$ for all quadrats was 1.14 and indicated a tendency towards an even distribution of entrances (range 0.91–1.44 m for individual quadrats).

The minimum nearest neighbour distances for each burrow were used to repeat the analysis using only one measurement per burrow (single-sector) and facilitated a comparison of the outcomes of both a single- and two-sector analysis of the same data. Both analyses indicated the same trend in burrow distribution. The $r_{a1}$ (1.08 ±0.03 SE m) was significantly greater than the $r_{e1}$ (0.88 ±0.03 SE m; $z_R = 6.62$, $P < 0.0001$), suggesting evenness of dispersion. Similarly, the observed $R_1$ for the single-sector analysis ($R_1 = 1.23$) tended towards the maximum possible ($R_{1.max} = 2.15$), although not as convincingly as the two-sector analysis.

The correlation coefficients between burrow density, two-sector divergence from randomness ($R_2$), distance to the edge of the colony, and distance to the edge of the island are presented in table 1 (the correlation between distance to edge of colony and distance to edge of island has little meaning and was not calculated). The only significant correlation was that between burrow density and $R_2$ ($r = 0.42$, $P = 0.04$), indicating that burrow entrances are more evenly dispersed at higher densities. All other correlations were not significantly different from zero; there was no linear relationship between the distance to the edge of the colony or the distance to the edge of the island and either dispersion of burrows or burrow density.

## DISCUSSION

The spatial distribution of Wedge-tailed Shearwater burrow entrances within the Radar Reef colony on Rottnest Island was non-random and tended towards evenness. The same result was obtained from both single- and two-sector nearest neighbour analyses of the data, and, therefore, the results are comparable with the single-sector analyses of Dyer & Hill (1990, 1995).

**TABLE 1**
**Correlation coefficients between spatial properties of Wedge-tailed Shearwater burrows at Radar Reef, Rottnest Island**

| | Burrow density | Two-sector divergence from randomness ($R_2$) |
|---|---|---|
| Distance to edge of colony | –0.19 ($P$ = 0.378) | 0.12 ($P$ = 0.592) |
| Distance to edge of island | 0.11 ($P$ = 0.609) | 0.08 ($P$ = 0.717) |
| Two-sector divergence from randomness ($R_2$) | 0.42 ($P$ = 0.040) | – |

Probabilities in parentheses. N = 233 burrows (in 24 quadrats). Note that the correlation between distance to edge of colony and distance to edge of island was not calculated.

Dyer & Hill (1990) implicated social relationships as the cause of burrow entrance aggregation in Wedge-tailed Shearwater colonies on Heron and Masthead islands. They suggested that shearwaters actively positioned their burrows near to one another to facilitate vocal communication and social interaction, and that, in some cases, offspring may choose to construct burrows in close proximity to their own parents. Dyer & Hill (1990) did not consider that structural stability of the colonies was an influential factor. A colony of much higher burrow density (more than fourfold) on North Stradbroke Island was examined and revealed a more even distribution of burrow entrances (Dyer & Hill 1995), as was observed at Radar Reef in this study, yet the authors maintained that structural stability was not a causal factor. Instead they offered an alternative explanation based on social behaviour. They argued that because the North Stradbroke colony was only 12 years old, the social interactions that drive clustering of burrow entrances (as suggested by Dyer & Hill 1990) had not had time to develop.

The Wedge-tailed Shearwater colony at Radar Reef has been present since at least the 1950s (Saunders & de Rebeira 1993), yet it still demonstrated a tendency towards evenness of burrow entrance distribution. It had a similar mean burrow density to that of North Stradbroke Island (when studied by Dyer & Hill 1995) and more than three times that of Heron and Masthead islands (when studied by Dyer & Hill 1990). I consider that burrow entrance distribution is more likely to be structurally, rather than socially, driven on Rottnest Island. From a structural perspective, the even distribution of burrow entrances is likely to be the most stable and would reduce the instances of burrow entrance collapse and erosion (Butler 1995, Bancroft *et al.* 2005). Other potential physical drivers of burrow distribution (e.g., soil strength, ground debris and vegetation type) were not measured in this study; however, given the even distribution of burrows at Radar Reef, these are likely to either be homogenous throughout the colony or insignificant in their influence on burrow distribution.

A significant positive correlation was observed between burrow density and the divergence from randomness of their distribution ($R_2$) within the Radar Reef colony. Burrow entrances in areas of higher density are more evenly distributed, a confirmation of the finding of Dyer & Hill (1990). By definition, burrows will be closer at higher densities. In this situation burrowing further (than random) from a neighbour, in an even pattern, has a selective advantage because it is likely to improve burrow entrance stability and avoid burrow collapse that may result in reduced reproductive success (through adult, chick or egg mortality, or the degradation of a suitable breeding site). Ramos *et al.* (1997) found that shorter nearest neighbour distance (between burrow entrances) had a negative impact on the hatching success in Band-rumped Storm Petrels, *Oceanodroma castro* (Harcourt, 1851), but no effect in Cory's Shearwaters, *Calonectris diomedea* (Scopoli, 1769). The potential mechanisms of the impact were not discussed by Ramos *et al.* (1997). Empirical evidence as to the structural stability of aggregated or evenly dispersed burrow entrances is not available. A comprehensive study that examines physical colony characteristics, social relationship, burrow stability and reproductive success, simultaneously, would clarify these issues.

When considered with the findings of Dyer & Hill (1990, 1995), the results from Radar Reef suggest that there is a density-dependent trade-off between social benefit (aggregated burrow entrances) and colony stability (evenly distributed burrows). It appears that at lower burrow densities (e.g., less than 0.1 burrows $m^{-2}$; Dyer & Hill 1990) Wedge-tailed Shearwaters favour an aggregated distribution of burrow entrances that may reflect social interactions. At higher densities (greater than 0.3 burrows $m^{-2}$; Dyer & Hill 1995, this study), the birds construct their burrow entrances in a more even distribution that, most likely, reflects attempts to maintain the structural stability of the colony.

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# IS COLONY ATTENDANCE BY SHEARWATERS INFLUENCED BY BRIGHT MOONLIGHT OR INCLEMENT WEATHER?

by Christopher Powell, Stuart Bradley and Ron Wooller

(with four text-figures and one table)

**Powell, C., Bradley, S. & Wooller, R.** 2008 (31:x): Is colony attendance by shearwaters influenced by bright moonlight or inclement weather? *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 35–44. ISSN 0080-4703. Department of Biological Sciences and Biotechnology, Murdoch University, South Street, Murdoch, Western Australia 6150 (CP, SB, RW); Department of Agriculture and Food, PMB 50, Esperance, Western Australia 6450 (CP*). *Author and address for correspondence. Email: cpowell@agric.wa.gov.au

If nocturnal colony attendance is associated with avoidance of predators, fewer nests would be visited by parent shearwaters on moonlit nights, or nests would be visited later, than on dark nights. Additionally, if shearwaters foraged at night on certain prey species, feeds delivered to nestlings would be larger on dark nights. We found that feed size and the probability of a nest visit were both greater on moonlit nights, while food delivery after midnight was not. A minimal risk of predation concurs with the disregard for moonlight shown by Flesh-footed Shearwaters, *Puffinus carneipes*, but does not explain their apparent preference for it. While nocturnal foraging is unlikely, it is not clear why feeds were significantly larger on moonlit nights, or why colony attendance by this species is strictly nocturnal. Weather may impact the ability of seabirds to forage at sea, although this is difficult to confirm. It was likely that nest attendance by parent shearwaters would be lower if certain variables negatively influenced foraging, but such influences may not manifest themselves immediately. Nest visits were examined in relation to local weather conditions at time-lag intervals of up to seven days. While no significant cross correlations were determined between burrow entry and two wind-velocity variables, a significant positive correlation was determined when the study site experienced pronounced unseasonal rainfall events.

**Key Words: colony attendance, moonlight, procellariiforms, shearwaters, *Puffinus carneipes*, wind, rainfall, Woody Island.**

## INTRODUCTION

The day-to-day behaviour of pelagic seabirds is poorly known because of the difficulties associated with studying them at sea. Accordingly, most knowledge is derived from data collected at the colony when the birds return to breed. Several procellariiforms (albatross, shearwater and other petrel species) are reportedly less numerous at their breeding colonies on clear, moonlit nights than on overcast or otherwise dark nights (Warham 1960, Harris 1974, Imber 1975, Watanuki 1986). Others have been shown to exhibit different behaviour in arriving at their nests later on moonlit nights (Storey & Grimmer 1986, Klomp & Furness 1992). These observations have been interpreted as a response to increased risks of predation and/or disturbance, together with limited or generally ineffective means of defence (Corkhill 1973, Watanuki 1986, Furness 1987).

However, contrary to these findings, Manuwal (1974) observed fewer birds on very dark nights. Similarly, whilst Storey & Grimmer (1986) noted high burrow activity on darker nights, the incidence of aerial vocalisation and courting behaviour on the ground was very much reduced and comparable to activity levels on the brightest nights. Corkhill (1973) suggested that gulls preyed upon Manx Shearwaters, *Puffinus puffinus* Brunnich, 1764, more intensively on moonlit nights because visibility was better, but that predation was higher on very dark nights than on nights of intermediate illumination because shearwaters were disoriented in dark conditions and more often landed near gull nests by accident. Harris (1969) discussed the colony attendance of two similar-sized species of storm-petrel that nested on the Galapagos Islands; one nocturnal and the other diurnal. The same attendance behaviour has also been reported in Cory's Shearwater, *Calonectris diomedea borealis* (Scopoli, 1769), breeding at two Atlantic colonies (Hamer & Read 1987, Klomp & Furness 1992), whilst McNeil *et al.* (1993) emphasised that some species maintained nocturnal nest attendance even in the absence of predation. Such observations suggest that predator avoidance is not a universal explanation for the avoidance of bright moonlight. Alternatively, colony visitation and chick provisioning may be influenced by the effect of ambient light conditions on prey availability close to the surface of the sea (Imber 1975). Bioluminescence and diel-vertical migratory behaviour of certain cephalopods (squid) found in some procellariiform regurgitates have supported the argument that certain species feed at night; supposedly because such prey are only visible and accessible near the surface at this time (Imber 1973, Clarke *et al.* 1981, Weimerskirch & Wilson 1992). This implies that overall food delivery would be lower on moonlit nights because prey are less available, whereas the anti-predator hypothesis predicts that the proportion of nests visited would be lower, but the size of feeds no smaller (Klomp & Furness 1992).

Climatic impacts upon seabirds have undoubtedly helped shape their demography and life-history characteristics. While severe perturbations occasionally result in large-scale mortalities (Schreiber 2002), the continuous rebalancing of the Earth's heat load is often responsible for extreme local weather conditions, the full gamut of which must be encountered by wide-ranging seabirds, particularly those that forage in the Southern Ocean (Brooke 2004). Weather may have an impact on the ability of seabirds to forage due to direction and velocity of wind affecting flight, the clarity/turbidity of water affecting their ability to identify and capture prey, as well as larger scale effects upon prey behaviour and distribution (Dunn 1973, Taylor 1983, Sagar & Sagar 1989, Finney *et al.* 1999). Like many aspects of seabird biology, the effects of weather upon seabirds at sea are poorly known because most information is gleaned at the

breeding colony. Warham (1960) proposed that the general state of the weather preceding the night in question, would have some effect on the numbers of adult shearwaters that came ashore to feed their young. However, from his study of Short-tailed Shearwaters, *Puffinus tenuirostris* (Temminck, 1835), breeding in Tasmania, he found no correlations between rainfall, or wind strength and direction, and nightly burrow visitation by parents.

In this paper we make a similar investigation of the same weather variables at time-lag intervals of up to seven days, and compare nest visitation between moonlit and dark nights. Specifically, we investigated whether parent shearwaters avoided returning to the colony on moonlit nights, or whether they delayed their return until after the moon had set. Additionally, we examined the influence of moonlight on the sizes of feeds delivered to chicks.

## METHODS

### The study species, the field site and data collection

From 2000–03 the Flesh-footed Shearwater, *Puffinus carneipes* Gould, 1844, a procellariiform seabird, was studied in the centre of its western range at a small colony on Woody Island (33°58'S, 122°01'E), Western Australia. This shearwater breeds in burrows at three distinct locations on the island, where adult birds make landfall after dark. There are no snakes or indigenous mammals on Woody Island; the Black Rat, *Rattus rattus alexandrinus* (Linnaeus, 1758), being the only potential terrestrial predator. However, large predatory birds include the Australian Raven, *Corvus coronoides* Vigors & Horsfeld, 1827, Pacific Gull, *Larus pacificus* Latham, 1802, and White-bellied Sea Eagle, *Haliaeetus leucogaster* (J.F. Gmelin, 1788).

Shearwater burrows were monitored daily using knock-down barricades, from mid-October, when adult birds returned to the colony, through to early May the following year, when the last fledglings departed. Twenty chicks were weighed once-daily over the three months they spent in the nest (Powell *et al.* 2007). During each year, food delivery was calculated by weighing a randomly selected sub-sample of ten chicks, every three hours between dusk and dawn, over seven-night intervals, at different stages of the nestling period (Hamer *et al.* 1997, Phillips & Hamer 2000).

The first weighing session commenced before dusk each evening, and the final session, 12 hours later, after dawn the following morning. Food delivery was recorded only when the barricade was displaced, and the chick exhibited a mass increase between weighings. Displaced barricades coinciding with no mass increase were attributed to factors other than chick provisioning, and were excluded from the analysis. Total overnight food delivery equalled the sum of positive mass increments between successive weighings. Frequent weighing trials conducted during February and March 2000 and February, March and April 2003, coincided with either a full moon or near-full moon when the moon rose shortly after sunset, and set around sunrise the following day. Thus, during clear weather there was bright moonlight over the colony for all or most of the night. By contrast, during nights of inclement weather or heavy cloud (which were common), the moon was obscured and no direct moonlight reached the colony. Burrow visitation was examined to determine whether ambient light influenced colony attendance by adult shearwaters. Nights of full moon with obscured moonlight were preferred to dark nights at other phases of the moon, or diurnal moonrise, to exclude any unknown astronomical influences.

To investigate whether moonlight increased the tendency of parents to visit their nests later (Storey & Grimmer 1986, Klomp & Furness 1992), nest visitation before and after midnight was also examined. Further, in addressing the hypothesis that feeds would be smaller on moonlit nights, in line with Imber (1975), food delivery to nestlings was compared between nights of bright moonlight and nights of total darkness.

### Astronomical and climatic data

Dates of full moon, times of moonrise and moonset, and sunrise and sunset were obtained online (Geoscience Australia 2003), while daily rainfall and three-hourly wind observations were supplied by the Bureau of Meteorology. The Esperance Meteorological Office is located some 20 km from Woody Island, so data may not have reflected island conditions exactly. Because three-hourly observations often exhibited considerable variation over the course of a 24-hour period, wind data were analysed in terms of both mean and maximum velocity (in knots) recorded in each 24-hour period from midnight, as well as the mean angle of the wind. Wind angle refers to its vector (the direction in which the wind is moving), whereas wind direction may be confused with the direction from which it is coming (e.g., a "southerly" comes from the south).

### Statistical analysis of data

The means of several variables associated with food delivery were compared between moonlit and dark nights. To take into account the repeat-measures effect of feeds delivered to the same nest, burrow entry data were analysed using the logistic random-effect regression module of the Egret Statistical Package for Windows™ (CYTEL 1999). Burrow identity was used as the repeat grouping variable; and whether the nest was visited on a particular night and, whether a feed was delivered after midnight, were the binary dependent variables. Year and measuring period were included as factors to absorb annual and seasonal variation in the frequency of visits. Tests of the mass of food delivered between dark and moonlit nights were performed using the mixed-model module of the SPSS Advanced Statistical Package for Windows™ (Norušis 1994). Burrow identity was treated as the subject variable of the repeat-measures model; and year, month and day within measuring periods were treated as the repeated effects. Moonlit versus dark nights was treated as a fixed factor.

To calculate wind angle, it was necessary to transform vectors in degrees to radians, for calculation as polar coordinates, before re-transformation to allow presentation of data relative to degrees (Zar 1999). The Axum 5.0B Package for Windows™ (MathSoft 1996) was used for this purpose, whilst time-lag correlations and other statistical analyses were performed using the SPSS Advanced Statistical Package for Windows™ (Norušis 1994). Time-lag cross correlations were used to detect daily changes (transform difference = 1) in nest visitation as a result of rainfall and changes in wind conditions at lag intervals of up to seven days (i.e., whether rainfall or wind influenced daily changes in the proportion of nests visited for up to seven days before, and seven days after those particular weather conditions). Significance was

determined if the cross correlation function exceeded the confidence limits (± two standard errors from zero).

## RESULTS

### Moonlight, burrow visitation and food delivery

Although the main influx of shearwaters typically occurred shortly after sunset, observations at the colony, and mass increases recorded during consecutive weighing sessions, revealed that birds arrived at the nesting area throughout the night. Table 1 shows 23 nights in total, of full moon (± up to three nights) when bright moonlight shone over the colony, and full moon (± up to four nights) when the moon was obscured and the colony was in darkness. Also, shown are the times (corrected for local time) of sunrise and sunset, and moonrise and moonset (Geoscience Australia 2003), as well as observations of conditions at the colony.

Using logistic random-effect regression with burrow identity as the grouping variable, and including year and month as factors, we determined that there was significantly greater probability of burrows being visited on moonlit nights than on dark nights (deviance difference $\chi^2_1 = 8.923$, $p = 0.003$, odds ratio = 1.77). In other words: moonlight increased the ratio of the odds of a visit (as opposed to no visit) by a factor of 77%, all other factors being equal. Initially, there was also a significantly greater probability of meals being delivered after midnight on moonlit nights (deviance difference $\chi^2_1 = 10.169$, $p = 0.001$, odds ratio = 2.08). However, when the regression was restricted to burrow nights where a visit was recorded, and the dependent variable was simply the occurrence of a visit after midnight; the presence of moonlight was not significant; indicating that, overall, moonlight did not produce significant changes in the probability of a visit occurring after midnight rather than before. Finally, feed size (the mean mass of food delivered overnight) was greater, by a highly significant margin, on moonlit nights (81.2 g ± 7.2 standard error) than on dark nights (71.5 g ± 6.6 standard error) when analysed using a repeat-measures mixed model ($F_{2, 88} = 123.181$, $p = <0.001$).

### Rainfall and wind characteristics

A significant correlation between nest visitation and rainfall was determined in 2000, when two pronounced rainfall events occurred. In late February over 70 mm of rain fell in 24 hours, followed by a deluge of similar proportions over 48 hours, two weeks later in March. The second event

**TABLE 1**
**Dates of full moon during 2000 and 2003. Times of sunrise/sunset, moonrise/moonset and observations of ambient light at the colony**

| Moon | Date | Sunrise | Sunset | Moonrise | Moonset | Observations at colony |
|---|---|---|---|---|---|---|
| Waxing[3] | 17/2/2000 | 0528 | 1844 | 1658 | 0224 | Clear; bright moonlight (gone from colony by 0130) |
| Full | 20/2/2000 | 0531 | 1840 | 1918 | 0545 | Clear; bright moonlight (gone from colony by 0445) |
| Waxing[3] | 17/3/2000 | 0553 | 1808 | 1630 | 0221 | Clear; bright moonlight (gone from colony by 0120) |
| Waxing[2] | 18/3/2000 | 0554 | 1806 | 1712 | 0327 | Clear; bright moonlight (gone from colony by 0230) |
| Waxing[1] | 19/3/2000 | 0555 | 1805 | 1750 | 0433 | Clear; bright moonlight (gone from colony by 0030) |
| Waning[2] | 18/2/2003 | 0529 | 1842 | 1954 | 0628 | Clear; bright moonlight |
| Waning[3] | 19/2/2003 | 0530 | 1841 | 2026 | 0738 | Clear; bright moonlight after 2300 |
| Waxing[1] | 17/3/2003 | 0552 | 1809 | 1748 | 0404 | Clear before 2330; rain after |
| Waning[1] | 19/3/2003 | 0554 | 1806 | 1855 | 0626 | Clear |
| Waxing[2] | 14/4/2003 | 0614 | 1731 | 1616 | 0250 | Patchy cloud; moonlight |
| Waxing[1] | 15/4/2003 | 0614 | 1730 | 1649 | 0400 | Intermittent light cloud; moonlight |
| Full | 16/4/2003 | 0615 | 1729 | 1722 | 0511 | Clear; bright moonlight |
| Waxing[2] | 18/2/2000 | 0529 | 1843 | 1751 | 0330 | Dark; heavy cloud, no moonlight |
| Waxing[1] | 19/2/2000 | 0530 | 1842 | 1837 | 0437 | Dark; heavy cloud, no moonlight |
| Waxing[3] | 16/3/2000 | 0552 | 1809 | 1543 | 0116 | Thunderstorm; heavy cloud, no moonlight |
| Waxing[4] | 14/2/2003 | 0526 | 1847 | 1707 | 0200 | Overcast; only occasional moonlight |
| Waxing[2] | 15/2/2003 | 0527 | 1846 | 1756 | 0302 | Overcast, drizzle, no moonlight |
| Full | 16/2/2003 | 0529 | 1843 | 1840 | 0409 | Heavy cloud, rain, no moonlight |
| Waning[4] | 20/2/2003 | 0531 | 1840 | 2059 | 0847 | Thunder, rain, heavy cloud, no moonlight |
| Full | 18/3/2003 | 0553 | 1807 | 1822 | 0515 | Heavy cloud; moon intermittent after 2330 |
| Waning[2] | 20/3/2003 | 0555 | 1805 | 1929 | 0737 | Clear, but colony in darkness until after 2100 |
| Waning[3] | 21/3/2003 | 0555 | 1803 | 2004 | 0849 | Clear, but colony in darkness until after 2100 |
| Waxing[3] | 13/4/2003 | 0613 | 1733 | 1542 | 0141 | Heavy cloud before 2100; moonlight at 2345 |

Superscript refers to the number of nights before or after full moon, e.g., Waxing[3] = three before, Waning[2] = two after.

caused flooding of burrows in low-lying areas, and resulted in some nestlings evacuating their burrow. Interestingly, rather than nest visitation being impeded, burrow attendance was 100%, 85% and 100% on the three respective dates of high rainfall. With the assumption that the March deluge was a single continuous event, the overall correlation for rainfall was significant and, not surprisingly, positive, at zero. In 2003, however, a single heavy downpour of 28 mm occurred two days after the lowest proportion of burrow entries (11%), which created a negative lag (fig. 1). In other words, there were fewer birds in burrows two days before rainfall. Although no significant cross correlations were determined between mean or maximum wind velocity (fig. 2), this rainfall event coincided with the highest recorded wind velocity (Day 104, 2003; fig. 3).

No significant correlation was determined between nest entry and mean wind angle (fig. 1), exemplified by polar plots in concurrence with maximum wind velocity (fig. 4). It was expected that, if significant correlations existed in relation to mean angle, circles would be concentrated in the outer radii (indicating higher proportions of visitation) in favoured wind conditions and, vice versa, in less-favourable conditions. Furthermore, if any significant interactive relationships existed between wind angle and wind velocity, the circles would be expected to vary in size with their placement upon polar coordinates in relation to favoured wind conditions. This was not the case in any single year. In a global plot encompassing all three years examined, a higher concentration of circles in the northerly quarter (315°–45°, fig. 4) reconciles with the normal pattern of prevailing southeasterly summer winds, punctuated by southwesterly changes.

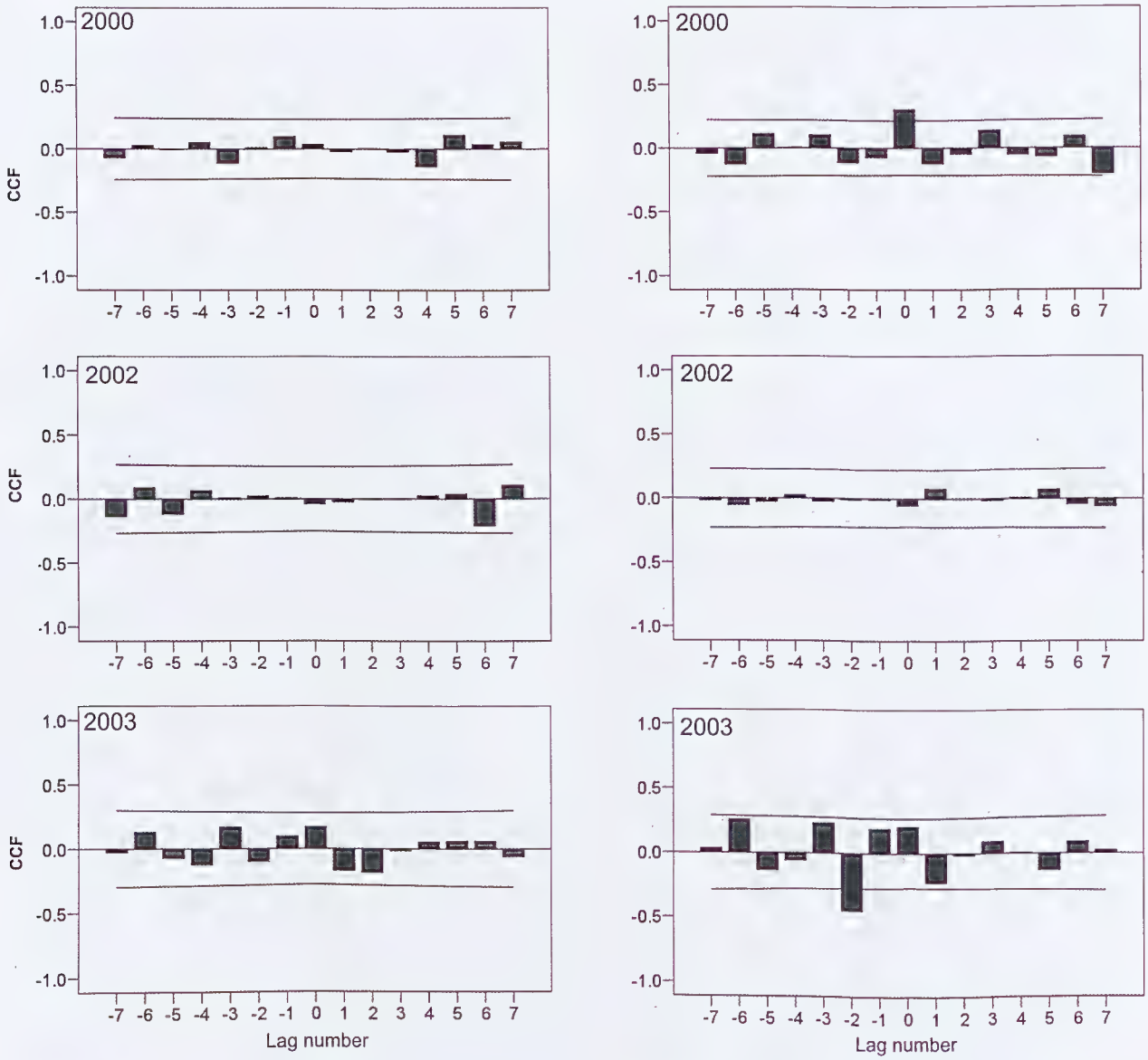


*FIG. 1 — Time-lag cross correlations (± 7 days) of daily changes in rainfall (right column) and mean wind angle (left column), with daily changes in the proportion of nests entered overnight. A significance correlation is determined when the cross correlation function (CCF), indicated by the black bars, exceeds the confidence limits (± two standard errors from zero). A significant positive correlation at zero days was identified for rainfall in 2000 (top right), and a significant negative correlation at -2 days (bottom right).*

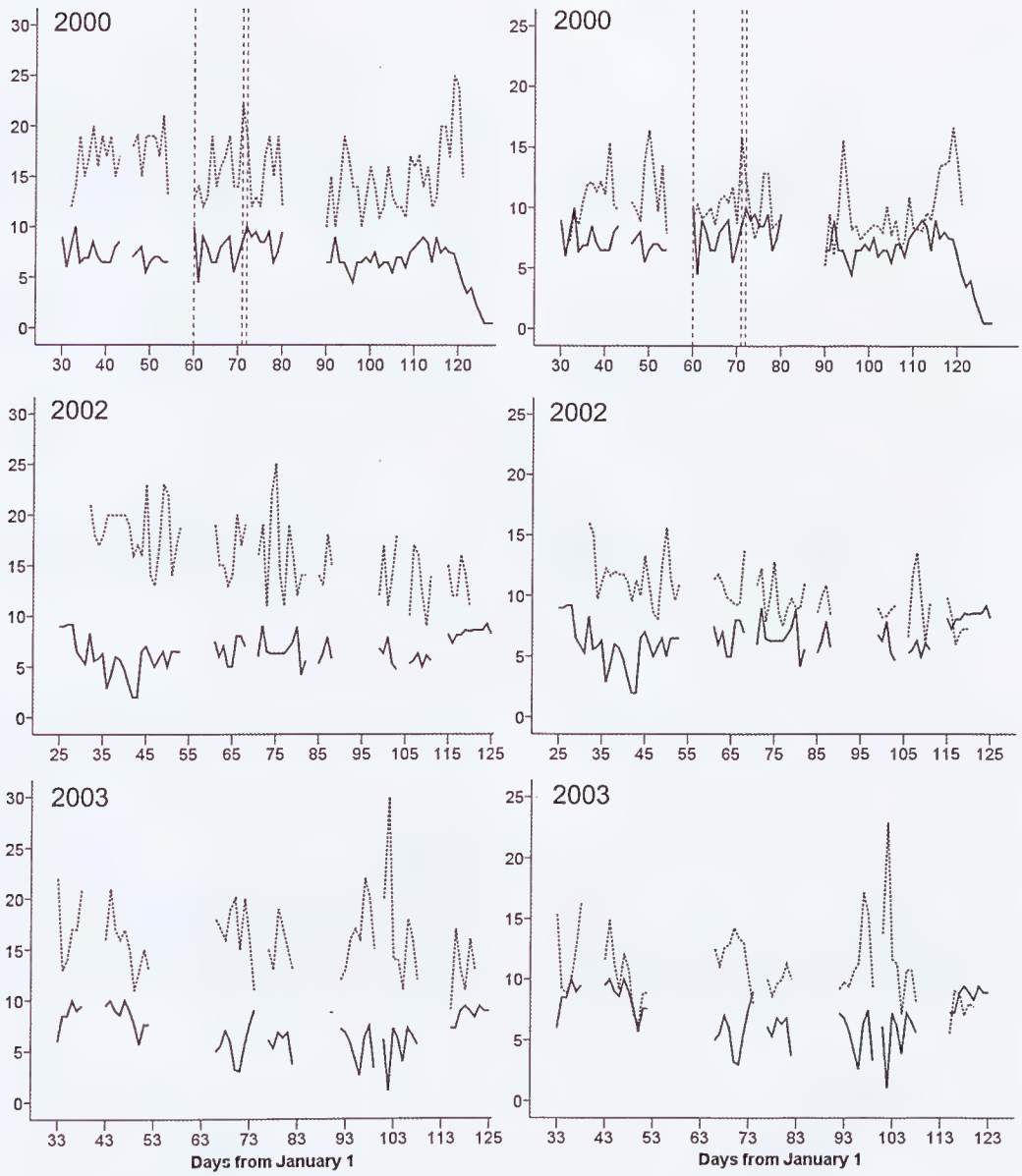


*FIG.2 — Time-series plots of the proportion (multiplied by a factor of 10) of nests entered overnight (bold line) in relation to mean (right column) and maximum (left column) wind velocities (broken line) in knots. The broken vertical lines on the two figures for 2000 denote significant rainfall events during that year.*

## GENERAL DISCUSSION

We found no negative influence of moonlight upon colony attendance by Flesh-footed Shearwaters. To the contrary, there was a greater probability of nests being visited on moonlit nights; feeds were larger and moonlight did not change the probability of a nest being visited after midnight rather than before. Similarly, we found no negative correlations between colony attendance and inclement weather conditions. Again, to the contrary, there was a significant positive correlation between burrow visitation and heavy rainfall.

Warham (1958) reported heavy egg-loss by Flesh-footed Shearwaters on Eclipse Island to the King's Skink *Egernia kingii* (J.E. Gray, 1838). However, this reptile does not occur on Woody Island and, it is not known whether its smaller congener, the Southwestern Crevice Skink, *E. napoleonis* (Gray, 1838), preys upon eggs, even though it was known to scavenge old ones. Similarly, the impact of the introduced Black Rat is unknown; but the large-bodied adult Flesh-footed Shearwater (625 g ± 3, Powell *et al.* 2007) is unlikely to be threatened. Both snakes and indigenous mammals are absent from the island, but Australian Ravens occasionally dug chicks from their burrows (Johnstone *et al.* 1990). The large predatory Pacific Gull occurs as only a few breeding pairs, and the White-bellied Sea Eagle is but an occasional visitor. Thus, predation of adult Flesh-footed Shearwaters on Woody Island is likely to be low or negligible.

Nightly observations at the colony from 2000–03 confirmed that many shearwaters arrived to feed their young around dusk or shortly thereafter. However, overnight weighing trials determined that although chicks received almost one meal per night on average (Powell *et al.* 2007) on many occasions both parents delivered food during the same night and, meals were delivered both before

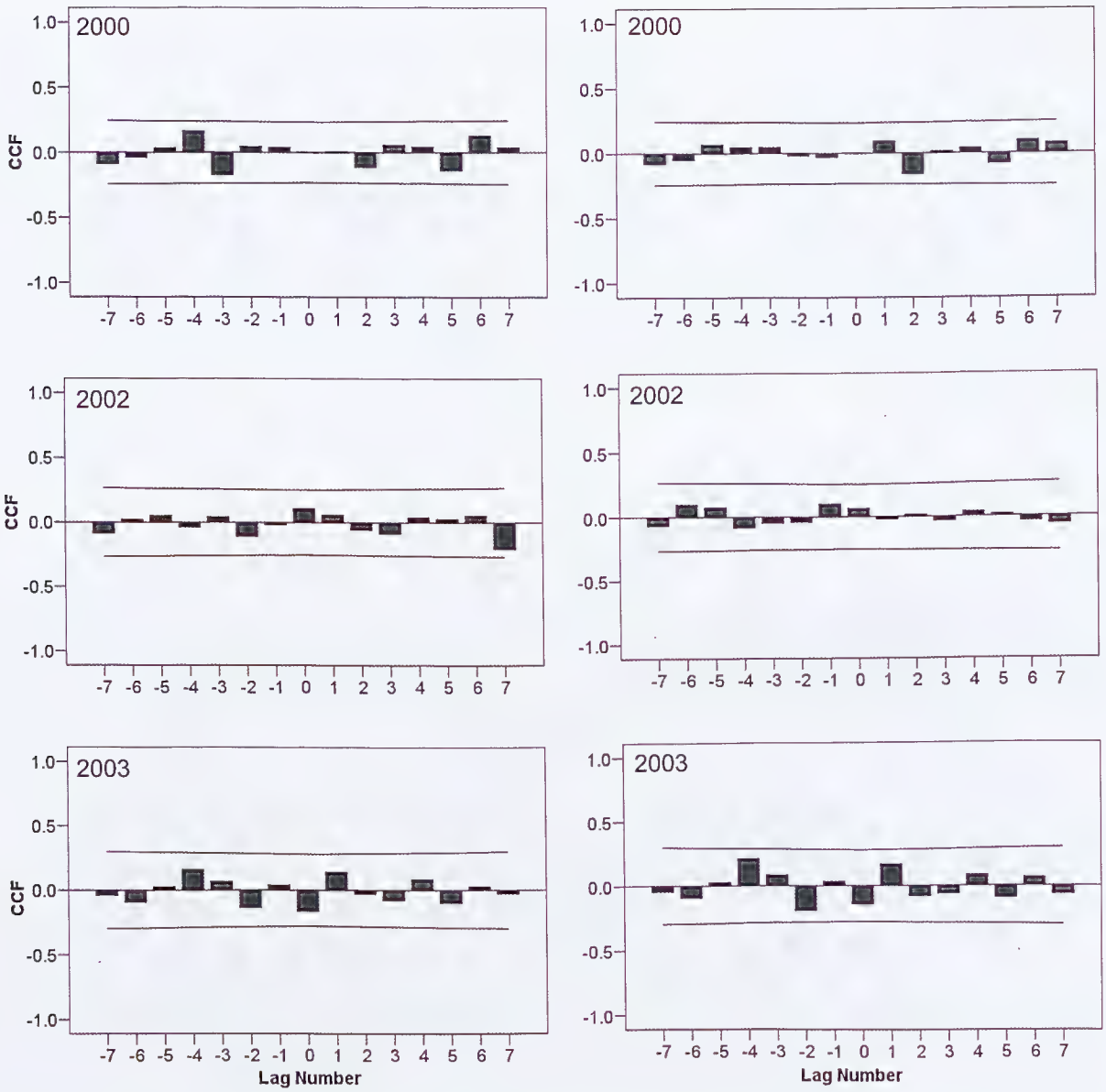


*FIG. 3 — Time-lag cross correlations (± 7days) of daily changes in mean (right column) and maximum (left column) wind velocities, with daily changes in the proportion of nests entered overnight.*

and after midnight. If shearwaters were less numerous at Woody Island on clear moonlit nights, it would support the hypothesis that moonlight avoidance is a response to an increased risk of predation (Corkhill 1973, Imber 1975, Watanuki 1986, Storey & Grimmer 1986), in line with the generally nocturnal colony behaviour of *Puffinus* shearwaters (Warham 1990, McNeil *et al.* 1993). A lower proportion of burrows visited would, therefore, be expected on nights of bright moonlight than on overcast or otherwise dark nights (Klomp & Furness 1992). Similarly, if adults were moonlight-shy there would be a greater tendency to delay returning to the colony until after moonset (Storey & Grimmer 1986, Klomp & Furness 1992). In this study, nights of full moon or near-full moon with moonlight obscured were examined, rather than dark nights at other phases of the moon, to exclude unknown astronomical factors that might influence or affect prey distributions or the behaviour of the birds themselves.

Using logistic random-effect regression software designed to absorb repeat-measures effects, as well as annual and seasonal variation, we determined that on moonlit nights there was a significantly greater probability overall, of burrows being visited and, initially, a significantly greater probability of food being delivered after midnight. However, the presence of moonlight was not a significant predictor of a burrow visit occurring after midnight when the regression was restricted to burrow nights where a visit was recorded. This indicated that, overall, moonlight did not produce significant changes in the probability of a visit occurring after midnight rather than before. Flesh-footed Shearwaters, therefore, demonstrated no moonlight avoidance behaviour, which corresponds with the notion that predation on Woody Island is low or negligible. By contrast, shearwaters showed a preference for conditions of ambient illumination at the colony — possibly because it enhances visual cues.

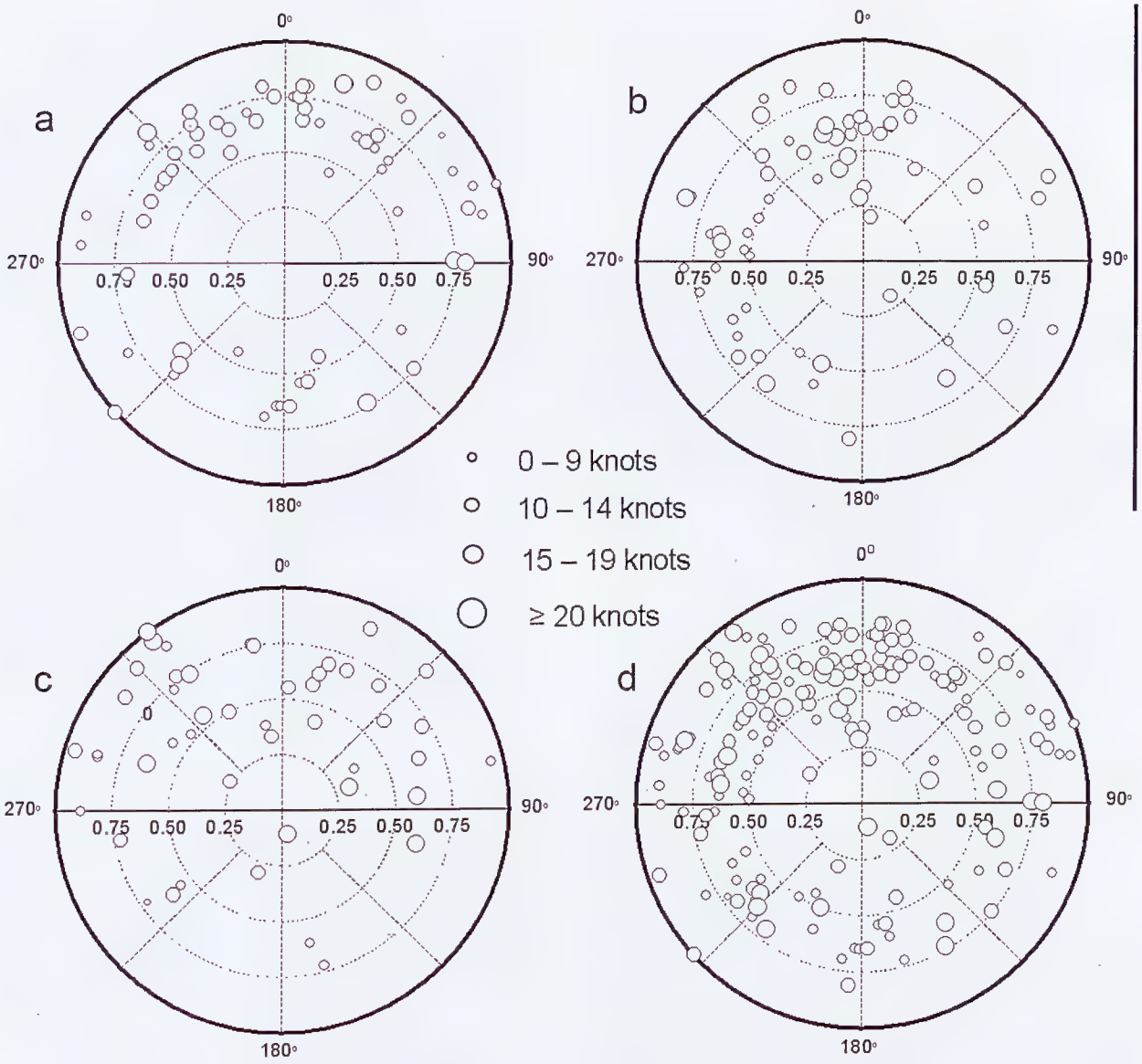


*FIG. 4 — Polar plots of the nightly proportion of nests entered during (a) 2000, (b) 2002 and (c) 2003, and (d) a global plot incorporating all three years, in relation to mean wind angle and maximum wind velocity. Proportions are defined by radial lines from the centre (zero), mean angle (direction of wind movement) is relative to degrees (0°–360°), and wind velocity is classified in four groups identified by open circles of increasing size.*

Further to this, a repeat-measures mixed-model analysis determined that feed size (the mean mass of food delivered overnight) was greater, by a highly significant margin, on nights of bright moonlight than on dark nights. The suggestion that some procellariiforms feed at night has been supported by the fact that certain cephalopods found in seabird regurgitates, would be visible and accessible to foraging birds only when they rise to near the sea surface at night (Clarke *et al.* 1981, Weimerskirch & Wilson 1992). Moreover, Gould (1967) observed Wedge-tailed Shearwaters, *Puffinus pacificus* (J.F. Gmelin, 1789), foraging at night, and Harper (1972) stated that Thin-billed Prions, *Pachyptila belcheri* (Matthews, 1912), fed mainly at night; on vertically migrating prey. Similarly, Grey-faced Petrels, *Pterodroma macroptera* (A. Smith, 1840), took 80% or more prey at night, of which about 90% consisted of bioluminescent cephalopods (Imber 1973, 1975). While Cory's Shearwater chicks were fed mainly on teleost fish, they also received cephalopods; and these were light-sensitive vertical migrants that rise close to the surface only on dark nights. Consequently, meals were significantly smaller on nights of full moon, suggesting that Cory's Shearwater foraged nocturnally and, with greater success on dark nights due to a greater availability of prey (Klomp & Furness 1992). In direct contrast, feeds delivered to Flesh-footed Shearwater chicks were of significantly greater mass on moonlit nights. Whilst detailed dietary information for this species is scarce, a stable isotope analysis of eggshell membranes determined a slight enrichment of $\delta^{13}C$, and a $\delta^{15}N$ enrichment of around 4‰ — evidence that during the pre-laying period at least, the dominant prey was pilchard *Sardinops sagax* (Steïndachner, 1879), a schooling teleost (J.N. Dunlop, unpubl. data). Similarly, intense fishery by-catch observations undertaken during March and April in 2007 and 2008 confirmed that scavenging by Flesh-footed Shearwaters around pilchard fishing vessels was most intense

at the time that chicks attained peak body mass (C. Powell, unpubl. data). Nocturnal foraging on vertically migrating prey is, therefore, unlikely, but the effects of moon phase on organisms at lower trophic levels, that may in turn affect the feeding of pilchards, are unknown. A more detailed investigation will be required to provide an explanation as to why feeds were significantly larger rather than not different.

Evolution in oceanic seabirds has resulted in extremely efficient operation in the environment upon which they depend for their food. For example, long-distance foragers such as albatross (*Diomedea* spp.) rely on persistent winds to fully exploit their given mode of flight (Tickell & Gibson 1968). Given the not-dissimilar flap-gliding flight of shearwaters (Pennycuick 1987), it was not surprising that higher wind velocities did not result in lower proportions of nest attendance. Although the day of the greatest mean and maximum velocities (23 and 30 knots) occurred two days after the lowest recorded burrow entry rate (11%), no significant correlation was determined overall. It had been expected that calm weather would result in fewer birds coming ashore, but this was not the case. While completely calm days were rare, the lowest mean wind velocities recorded (5–6 knots) coincided with nest visitation rates of 65–80%. If, indeed, this population can feed relatively close to the colony, then wind energy as a component of the energy budget of prolonged foraging trips may not be as crucial as it appears for some species, and meal delivery frequency would remain high even over periods of low wind velocity. However, low burrow attendance two days preceding inclement weather events may indicate changes in foraging behaviour in anticipation of the event.

The lack of correlation between nest visitation and mean wind angle is congruent with the observation that aspect was not a significant determinant of colony location (Powell *et al.* 2007). The northeasterly aspect of one of the three breeding areas suggests that colonised areas are not necessarily selected in relation to the seasonal prevailing wind. Moreover, the term "mean wind angle" is itself somewhat arbitrary. By definition, it represents the calculated mean of eight three-hourly observations, which may have varied considerably over the course of a 24-hour period. Each, however, is of equal importance because temporal wind conditions while adults forage at sea are likely to have a greater influence upon their foraging success than the immediate conditions at nightfall and, consequently, whether or not they return to the colony. The high concentration of nest visits that coincided with winds from the southeasterly to southwesterly quarter are likely to be a consequence of the prevailing southeasterly winds during the summer months, which are punctuated by southwesterly changes.

We conclude that Flesh-footed Shearwaters did not demonstrate moonlight avoidance behaviour at their breeding colony. To the contrary, they displayed some preference for moonlit nights, when burrows were more likely to be visited and feeds were significantly larger. Although, burrow visitation was high during pronounced rainfall events when the correlation was significant at zero, it is likely that shearwaters anticipate approaching weather and modify their foraging behaviour accordingly. Otherwise, nest visitation was not significantly influenced by rainfall, or by wind direction or velocity, at time-lag intervals of up to seven days preceding or following the event.

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(accepted 5 August 2008)

# SEABIRD BYCATCH IN NEW ZEALAND TRAWL AND LONGLINE FISHERIES, 1998–2004

by S. M. Waugh, D. I MacKenzie and D. Fletcher

(with nine text-figures and 17 tables)

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Fisheries bycatch is a threatening process for populations of procellariiform seabirds, and is of particular importance for the conservation of albatross, an especially threatened group at a global scale. There is a high level of endemism of albatross and petrels in New Zealand waters, and around one-third of the world's species of procellariiform seabirds breed in this area. Therefore, understanding the levels of mortality of these species in the New Zealand Exclusive Economic Zone is important for global conservation of the order. For New Zealand fisheries for the 1998–2004 fishing years, we estimated total seabird bycatch using data from scientific observers with model-based estimation procedures. Although sectors of the fishing activity were not evenly covered by observers, we were able to estimate seabird bycatch for large-scale fisheries by vessel size (split at 28 m length), season, area and year. Approximately 5500 seabirds (credible interval between 2000 and 10 000) are estimated to be landed in New Zealand trawl and longline fisheries annually, as a result of interactions with fishing gear. Few data were available for the small vessels, thus estimates are highly uncertain. Mortalities are likely to be most common in trawl fisheries at approximately 2000–3000 seabirds annually, with the greatest contribution coming from large vessels. Around one half of these birds were albatross. For large surface longline vessels we estimated that fewer than 500 seabirds were killed annually during the main tuna fishing season. For large demersal vessels, seabird mortality was estimated to have decreased from around 1800 seabirds in 2001 to 600 seabirds in 2004. We report observed captures by species for each fishing method and area for the fishing years 1998–2004. Thirty-one species of Procellariiformes were identified during this period, over half of which are threatened species. For some species, such as White-chinned Petrel, *Procellaria aequinoctialis* and White-capped Albatross, *Thalassarche steadi*, several hundred individuals were caught. For 15 species, fewer than 10 individuals were identified. However, the unrepresentative deployment of observer coverage across fishery areas makes it difficult to interpret the conservation implications of species captures. A high proportion of the petrel species was observed caught primarily from areas surrounding their breeding sites while albatross were caught across breeding and non-breeding areas. Greatly improved observer sampling ratios, and studies of population status and trends, are needed to understand the conservation implications of the effects of New Zealand trawl and longline fishing mortalities on seabird populations.

**Key Words: albatross, petrel, mortality, New Zealand, conservation, Procellariiformes, White-chinned Petrel, White-capped Albatross, model-based population estimates, *Procellaria aequinoctialis*, *Thalassarche steadi*.**

## INTRODUCTION

Estimating seabird mortality in fisheries is a key factor in assessing the sustainability of fisheries (Small 2005, Lack 2007). Fisheries bycatch has been identified as a threat to most albatross species (Gales 1998), and also results in the deaths of a large number of petrels (Brothers *et al.* 1999). Supplementary feeding of seabird populations on fisheries waste can boost population numbers for some species, as documented for the faster-breeding Lariidae species in the northern hemisphere (Furness 2003). However, the resulting complex changes in seabird community structure can be deleterious, as slower-breeding or more vulnerable populations of seabird can suffer increased predation pressure and competition as a result of increases in some larger predatory gull populations (Furness 2003). Fisheries mortality is identified as an important driver for population decreases for the slower-breeding albatross and petrel populations (Croxall *et al.* 1990, Weimerskirch *et al.* 1997, Tuck *et al.* 2001, Cuthbert *et al.* 2003, Nel *et al.* 2003, Rolland *et al.* 2007), while climate effects and colony-based impacts have more recently been recognised as contributory factors in population changes (Barbraud &Weimerskirch 2003, Rolland *et al.* 2007).

Recent reviews show rapidly worsening conservation status for albatross populations. The International Union for the Conservation of Nature (IUCN) defines albatross (family Diomedeidae) as the most threatened family of birds in the world. In 2008, all of the 22 albatross species are listed as Threatened or Near Threatened, three are Critically Endangered, and seven are Endangered (IUCN 2008). The family Diomedeidae is becoming threatened at a faster rate than any other in the world. Petrels (family Procellariidae) are less well studied, and of 80 extant species listed by the IUCN in 2008, 35 are classified as threatened. Albatross and petrel species are particularly vulnerable to population effects of fisheries mortality due to their demographic characteristics, with delayed breeding, low fecundity (Hamer *et al.* 2002) and migratory habits which take them into areas where they are exposed to fishing mortality throughout large parts of their life-cycles (de L. Brooke 2004, Small 2005).

There is international recognition of the importance of reducing seabird bycatch in fisheries, primarily for surface longline fishing (Brothers *et al.* 1999). This was the focus of the Food and Agriculture Organisation of the United Nations International Plan of Action for Reducing the Incidental Mortality of Seabirds in Longline Fisheries (FAO 1999). The importance of mortality of seabirds in trawl fisheries has been recognised as a problem for species conservation (Bartle 1991, New Zealand Ministry of Fisheries and Department of Conservation 2004, Sullivan *et al.* 2006).

Over one-third of the 104 species in the order Procellariiformes (albatross, petrels, storm- and diving-

**TABLE 1**
**Species of seabird observed caught in New Zealand fisheries in the 1998 to 2004 fishing years, returned by scientific observers and formally identified to species level**

| Species | Scientific name | IUCN threat status | Breeding distribution | FMA1 | FMA2 | FMA3 | FMA4 | FMA5 | FMA6 | FMA7 | FMA8 | FMA9 | FMA10 | Total observed caught 1997–2004 | % individuals caught in breeding areas |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| White-chinned Petrel | *Procellaria aequinoctialis* | VU | W | 1 | 5 | 26 | 357 | 329 | 218 | | | | | 936 | 23 |
| White-capped Albatross | *Thalassarche steadi* | NT | NZ | | 5 | 27 | 4 | 456 | 275 | 32 | | | | 799 | 34 |
| Sooty Shearwater | *Puffinus griseus* | NT | W | | 1 | 257 | 29 | 326 | 59 | | | | | 681 | 99 |
| Grey Petrel | *Procellaria cinerea* | NT | W | 13 | 9 | 7 | 35 | | 326 | | | | | 471 | 69 |
| Salvin's Albatross | *Thalassarche salvini* | VU | NZ | | 16 | 31 | 56 | 14 | 114 | | | | | 231 | 55 |
| Buller's Albatross | *Thalassarche bulleri* | VU | NZ | | 1 | 19 | 19 | 9 | 6 | 34 | | | | 178 | 16 |
| Antipodean Albatross | *Diomedea antipodensis* | VU | NZ | | 68 | 1 | | 2 | | | | | | 71 | 0 |
| Campbell Albatross | *Thalassarche impavida* | VU | NZ | 2 | 3 | 1 | 1 | 3 | 1 | 8 | | | | 46 | 2 |
| Flesh-footed Shearwater | *Puffinus carneipes* | LC | W | 19 | 16 | | | | | | | | | 35 | 54 |
| Short-tailed Shearwater | *Puffinus tenuirostris* | LC | W | | | | | 33 | | | | | | 33 | 0 |
| Cape Pigeon | *Daption capense* | LC | W | | 2 | | 8 | 7 | 7 | 5 | | | | 29 | 48 |
| Grey-faced Petrel | *Pterodroma macroptera* | LC | NZ | 4 | 3 | | 1 | | | | | | 1 | 18 | 39 |
| Black-browed Albatross | *Thalassarche melanophris* | EN | W | | 8 | 1 | 2 | | 3 | 2 | | | | 16 | 19 |
| Southern Royal Albatross | *Diomedea epomophora* | VU | NZ | | | 2 | 2 | 1 | | | | | | 14 | 14 |
| Northern Giant Petrel | *Macronectes halli* | NT | W | | 1 | 1 | 7 | | 3 | | | | | 12 | 83 |
| Chatham Albatross | *Thalassarche eremita* | CR | NZ | | | | 11 | | | | | | | 11 | 100 |
| Black Petrel | *Procellaria parkinsoni* | VU | NZ | 9 | | | | | 3 | | | | | 9 | 100 |
| Common Diving-Petrel | *Pelecanoides urinatrix* | LC | W | | | | | 5 | 3 | | | | | 8 | 100 |
| Southern Giant Petrel | *Macronectes giganteus* | NT | W | | 2 | | | | 4 | | | | | 6 | 0 |
| Fairy Prion | *Pachyptila turtur* | LC | W | | | | | | | 6 | | | | 6 | 100 |
| Westland Petrel | *Procellaria westlandica* | VU | NZ | | 1 | | | 1 | | 2 | | | | 4 | 50 |
| Light-mantled Albatross | *Phoebetria palpebrata* | NT | W | | | 1 | | | | 2 | | | | 3 | 0 |
| Northern Royal Albatross | Diomedea sanfordi | EN | NZ | | | | 2 | | | | | | | 2 | 100 |
| White-headed Petrel | *Pterodroma lessonii* | LC | W | | | | | | | | | | 2 | 2 | 0 |
| Buller's Shearwater | *Puffinus bulleri* | NT | NZ | 2 | | | | | | | | | | 2 | 100 |
| Fluttering Shearwater | *Puffinus gavia* | LC | NZ | 2 | | | | | | | | | | 2 | 100 |
| Grey-backed Storm-Petrel | *Garrodia nereis* | LC | W | | | | | | 2 | | | | | 2 | 0 |
| Antarctic Prion | *Pachyptila desolata* | LC | W | | | | 1 | | 1 | | | | | 2 | 50 |
| White-faced Storm-Petrel | *Pelagodroma marina* | LC | NZ | | | | 2 | | | | | | | 2 | 100 |
| Black-bellied Storm-Petrel | *Fregetta tropica* | LC | W | | | | | 1 | | | | | | 1 | 0 |
| Broad-billed Prion | *Pachyptila vittata* | LC | W | | | | | 1 | | | | | | 1 | 100 |

The IUCN status (IUCN 2008) and breeding areas (shading indicates breeding locations after Taylor 2000a, b) within Fishery Management Areas (FMA) are provided. Few birds were observed caught in FMA8-10, but little observer coverage was deployed in these areas. New Zealand endemic species (NZ) and species with world-wide breeding distribution (W) are indicated. The percentage of individuals returned following capture in areas where breeding colonies exist is shown (% individuals caught in breeding areas). Species are ranked by the number observed caught.

petrels) breed in New Zealand waters, including 12 of the 22 species of Diomedeidae. There is a high level of endemism among these species in New Zealand waters, with 12 endemic petrels (Procellariidae) and eight endemic albatross (Wilson 2004; table 1). This places considerable responsibility for the conservation of these species in the hands of the managers of the natural resources of New Zealand.

Understanding fisheries effects on populations is complex, as many of the key variables required to examine causal aspects of the relationships cannot be estimated due to the paucity of fisheries bycatch data and uncertainty around environmental influences on populations. Further uncertainty is introduced by the effect of fisheries mortalities out of the breeding areas, such as during the extensive inter-breeding migrations of albatross and petrels.

For New Zealand fisheries, less than 5% of total fishing effort was observed by scientific observers on average during the fishing years 1998–2004. Few fisheries have sufficient observer coverage to allow precise estimation of seabird bycatch. Observer coverage has been particularly poor for small vessels. Not all individuals killed or injured in fishing operations are recovered on board, so there is an underestimation of the total removals of seabirds from populations from observer data.

In this study, we sought to provide a comprehensive analysis of seabird bycatch for all New Zealand fisheries using surface longline (SLL) and bottom or demersal longline (BLL) and trawling (TR) methods. Previous studies of bycatch in New Zealand fisheries have focused on specific target fisheries (Bartle 1991, Murray *et al.* 1993, Baird 2004a, b, c, 2005, Baird & Smith 2007), and do not provide comparative data across the New Zealand Exclusive Economic Zone. The results reported here provide a partial examination of the fishing mortality in New Zealand waters, due to lack of data for some fishing methods, and our inability to include seabirds not landed as part of the fishing operation.

Trawl and longline fishing methods are those recognised as the most likely to catch seabirds in New Zealand waters (New Zealand Department of Conservation and Ministry of Fisheries 2004). Set net (gill net) fisheries may contribute substantially to seabird bycatch (Furness 2003, New Zealand Ministry of Fisheries 2007b), but due to extreme data paucity for these fisheries, data from this fishing method were not included in the current analysis. We used Bayesian methods to predict incidental capture likelihoods, and to identify the regions around the New Zealand Exclusive Economic Zone that are important for incidental mortality of seabirds, and in particular albatross.

## MATERIALS AND METHODS

Material presented in this study was reported to the Ministry of Fisheries (MacKenzie & Fletcher 2005).

### Data

Data were extracted from the New Zealand Ministry of Fisheries databases for TR, SLL and BLL fisheries for fishing events from 1 October 1997 to 30 September 2004. From these, we examined characteristics of the fishing events in the sample (e.g., start time, location, sea and weather conditions) and the number of captures of each observer-identified seabird species. Fishing effort (the number of fishing events) is described in units of tows for trawl fisheries. The unit of sets was chosen for longline fisheries as we considered that the data provided more consistency this way, with the number of hooks being a characteristic of a particular set. Therefore these data were used to estimate bycatch rates per unit of effort.

To predict total seabird bycatch, datasets that described the total fishing effort were accessed which were then scaled to total effort to give estimated mortality. Data were checked for obvious errors and omissions (e.g., statistical area or Fishery Management Area (FMA) not recorded) and corrections made where possible.

Fishing years were described for the 12-month periods from 1 October–30 September, named for the latter year in each split-year period. For example, the 1998 fishing year was the period 1 October 1997–30 September 1998.

Initial assessments of the data indicated that it would not be practical to estimate bycatch at the level of individual seabird species; hence the data were pooled for all species. However, for trawl fisheries, bycatch of albatross species was separately estimated. No attempt was made to differentiate between live and dead landed birds due to small sample sizes, or to allow for the possibility that birds released alive may have their long-term survival compromised as a result of injuries sustained during capture. Bycatch was therefore described as the number of individuals caught during fishing operations. These figures do not include individuals that may have struck the vessel superstructure and been found on the vessel as a result. But these birds do contribute to the total mortality caused by the fishing fleet, as do birds striking trawl cables and other non-observed fishing gear-related deaths.

The ten FMAs designated for New Zealand fisheries are shown in figure 1. When grouped these areas correspond roughly to biogeographical regions, with tropical petrel species' breeding areas concentrated in FMA1–2 and FMA9–10, and temperate petrels' and albatross' breeding sites in FMA4–6. Data on individual seabirds captured in New Zealand fisheries

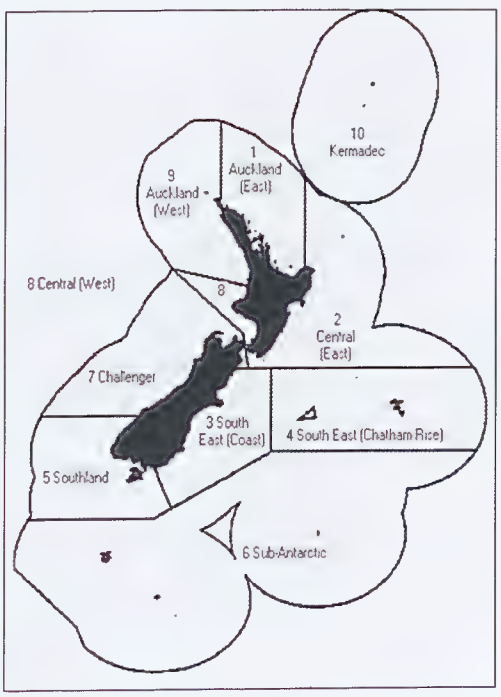


*FIG. 1 — Fishery Management Areas (FMA) in the New Zealand Exclusive Economic Zone.*

were extracted from Ministry of Fisheries databases together with summary information about the location of capture (FMA) and fishing method (TR, BLL and SLL). Around 4500 individual seabirds were reported captured by fisheries observers during the period of the study, of which 3641 were identified to species level by experienced seabird researchers ashore (methods reported in Bartle 2000, Robertson *et al.* 2004). We limited our examination of species data to those species formally identified caught in New Zealand fisheries (table 1). Thirty-one species were identified caught; their conservation status (IUCN 2008) and breeding areas in New Zealand waters were described, based on breeding sites identified by Taylor (2000a, b).

No statistical analyses are undertaken at species level due to the paucity of data, and unrepresentative nature of the observer sampling across the fishing effort. Results for species are reported by summing the captures for each species over the six-year period of the study.

## Statistical Methods

Seabird bycatch was estimated using a model-based method in order to account for some sources of variation in bycatch with auxiliary variables, and exploit similarities in bycatch rates among different seasons, years, regions or fisheries. For example, the model could be used to predict seabird bycatch in fisheries where there is little or no observer coverage, under the assumption that the model is a reasonable representation of the biological reality.

However, as for any estimation method, generalising results beyond the observed vessels requires the data to be collected within a statistical valid sampling scheme such as a random or representative sampling scheme. This is because there may be factors influencing bycatch that are not allowed for in the modelling. If the observed sets/tows are not representative of the whole fishery with respect to any of these unmodelled factors, the resulting estimates may be biased. Having as much randomisation in the sampling scheme as possible helps avoid the observed data being unbalanced in this way. The deployment of observers across New Zealand fisheries is neither random nor representative, as observers tend to be placed on larger vessels, and observations occur for a series of fishing events on a fishing trip, rather than being randomly distributed across all effort. The data were distributed across varying proportions of the fishing fleet and seasons, and one needs to be mindful of this when interpreting the results.

To account for the large number of fishing events with zero bycatch, a Poisson mixture model consisting of two components was used. The first component is a binary (0-1) random variable ($Z$) which is used to model the excess number of zeros. If $Z = 0$ then no seabirds will be caught, but if $Z = 1$, then the number of seabirds caught will be a random value from a Poisson distribution (the second model component). We therefore needed to estimate two parameters; the probability $Z = 1$ ($p$) and the Poisson rate parameter ($\lambda$; lambda). These parameters can be modelled to allow them to vary for fishing events conducted in different years, regions, seasons and so forth. Here, up to five factors were included in the models for $p$ and $\lambda$: fishing year, season, fishing area, target species and vessel size class. The seasons were defined to be spring (Oct–Dec), summer (Jan–Mar), autumn (Apr–Jun) and winter (Jul–Sep). Fishing areas were based upon FMAs and vessels were categorised by size, with large vessels being those of greater than 28 m total length.

The probability that $Z = 1$ for a fishing event in fishing year Y, season S, fishing area F by a vessel of size category V was modelled as

$$\text{logit}\,(\rho_{YSFV}) = \alpha + \beta_Y + \delta_S + \gamma_F + \upsilon_V$$

where $\beta$, $\delta$, $\gamma$ and $\upsilon$ denote the year, season, fishing area and vessel class effects respectively. Note that in order to fit this model one level of each of these effects had to be set equal to zero to establish a control, or baseline, situation where the model would reduce to logit ($\rho_{YSFV}$). For example, in trawl fisheries the "standard" situation was defined to be tows conducted in spring of the 1998 fishing year, in FMA7, by vessels longer than 28 m, and so the constraints $\beta_1 + \delta_1 + \gamma_7 + \upsilon_1 = 0$ were applied. The term "logit" simply denotes that the logit-link function was used to ensure values of $p$ remained on the 0–1 scale. This is the same link function used in standard logistic regression analyses. Biologically, this component could be regarded as modelling the probability that seabirds are at risk of being caught. This will be primarily driven by the seabirds rather than the fishers, and so target species was not included as a factor in this model. Vessel class was included here as a surrogate for effort (e.g., larger vessels may trawl for longer or have more hooks per set than smaller vessels).

The Poisson rate parameter was modelled using a log-link function (to maintain values of $\lambda$ in the range 0–infinity) with the same factors as above and the addition of target species. Hence, given seabirds were at risk of being caught, the bycatch rate for a fishing event in year Y, season S, fishing area F by a vessel of size category V targeting species T was modelled as,

$$\text{logit}\,(\lambda_{YSFVT}) = \alpha' + \beta'_Y + \delta'_S + \gamma'_F + \upsilon'_V + \tau'_V.$$

The model was fit to the data using Bayesian statistical methods (Markov chain Monte Carlo (McMC)) in the software WinBUGS. The use of McMC methods is becoming more widespread in many areas of applied statistics, including fisheries and wildlife ecology. In a Bayesian analysis a model parameter is considered a random variable. A *prior distribution* is defined for each parameter that represents the state of knowledge about that parameter prior to data collection. Here, vague or uninformative prior distributions were used for all model parameters (normal distributions with mean = 0.0 and variance = 100.0). The observed data were then used to update our knowledge of the model parameters, resulting in the *posterior distribution*. Until recently, the Bayesian approach to statistical inference has been limited in its applications due to difficulties in analytically estimating the posterior distribution for model parameters. However, McMC methods are an iterative, computer intensive method for approximating posterior distributions of parameters. These methods also provide a flexible analysis framework that can be simply applied to problems that are difficult to address with a non-Bayesian philosophy.

For the McMC analysis, three chains with different initial values were used for each model. Initial values were chosen by first using maximum likelihood to get point estimates of the parameters, then selecting a random value from a normal distribution around that point estimate with a standard deviation corresponding to a 30% coefficient of variation. The chains were initially run for 10 000 iterations then visually inspected for convergence and mixing. If it was determined that convergence had been achieved then

the chains were run for a further 25 000 iterations to estimate the posterior distributions of the model parameters (hence 75 000 samples per parameter), otherwise additional iterations were performed until convergence was achieved. To predict the observed and unobserved bycatch, 10 000 post-convergence iterations were used for trawl fisheries and 20 000 iterations for longline fisheries.

As part of the analysis, it is also very easy to predict the outcome of unobserved sampling units. For example in this context, in the summer of fishing year 2004, FMA6, trawl vessels longer than 28 m targeting squid (i.e., Y = 7, S = 2, F = 6, V = 1 and T = 6) conducted 2623 unobserved tows. Given the current estimates of the model parameters ($\hat{p}_{7,2,6,1}$ and $\hat{\lambda}_{7,2,1,6}$), the number of unobserved tows where seabirds are at risk of capture ($x$) could be predicted as a binomial random variable from 2623 trials with probability of "success" $\hat{p}_{7,2,6,1}$. Likewise, using the properties of Poisson random variables, the number of seabirds caught on unobserved tows would be a random value from a Poisson distribution with rate parameter $x\hat{\lambda}_{7,2,6,1,6}$. The total level of bycatch is then the predicted number plus the number actually observed. Predicting bycatch in this manner automatically accounts for two important sources of variation: uncertainty in the level of bycatch for specific values of $\hat{p}_{7,2,6,1}$ and $\hat{\lambda}_{7,2,1,6}$ (due to the stochastic nature of bycatch), and uncertainty in the values of $\hat{p}_{7,2,6,1}$ and $\hat{\lambda}_{7,2,1,6}$ themselves.

From the model, bycatch is predicted at a relatively fine scale (i.e., year × season × area × vessel class × target species). Here results are reported at coarser scales (e.g., year × vessel class) which were achieved by aggregating the predicted bycatch across the unreported factor levels. However, as the scientific observer program has tended to target larger vessels, results are reported separately for each vessel size class, as those for vessels smaller than 28 m may be less reliable.

The posterior distribution for the estimated level of bycatch is summarised by the median and the 2.5$^{th}$ and 97.5$^{th}$ percentiles (i.e., the central 95% credible interval).

To verify the reasonableness of the model, the above procedure for predicting bycatch on unobserved tows can also be applied to observed tows. That is, the model can be used to predict the number of seabirds caught for observed tows in each combination of year × season × area × vessel class × target species. If the model is reasonable, then the observed number of captures should lie within the "typical" range of predicted values (i.e., it would be expected that 95% of the observed values lie between the 2.5$^{th}$ and 97$^{th}$ percentiles of the distributions for the predicted values). Standardised residuals could also be calculated from the predicted distributions as:

$$R_i = \frac{O_i - P_i^{50}}{(P_i^{97.5} - P_i^{2.5})/4}$$

where $O_i$ is the observed number of seabirds caught in the $i^{th}$ combination of year, season, area, vessel class and target species and $P_i^k$ is the $k^{th}$ percentile from the distribution of the predicted number of seabirds caught (thus $P_i^{50}$ is the median of this distribution). The denominator is simply a between-combination scaling factor that has been used to place the residuals on a common scale, similar to that used in linear regression. Note however that these standardised residuals are not expected to be normally distributed because the data come from an extremely skewed, discrete-valued distribution. Thus residual Q-Q plots may not result in a straight line even if the model is a reasonable representation of the data. These residuals were plotted against various factors to check for systematic deficiencies within the model, as is typically done for linear regression applications.

### Examining risk to seabirds between methods and areas

To assess the risk of bycatch for different subsets of the New Zealand fishing methods, areas and seasons, we used the results from the bycatch modelling to estimate the *bycatch* (number of birds) and the *capture rate* (birds per set or per 100 tows) for TR, BLL and SLL fishing. We aggregated the predictions across target species. This allowed us to calculate an estimate of bycatch and capture rate for different vessel-method-area-season groups (fishery strata) combining different vessel sizes (small or large), method (TR, SL, BLL), fishing area (FMA1 to FMA10) and season (spring, summer, autumn and winter).

We report the results using estimates for the 2004 fishing year, which was similar across all areas to that for an average estimate across all fishing years (1998–2004 for TR and SLL; 1999–2004 for BLL). As 2004 was the most recent year included in our model, we considered it was therefore the most likely to be representative of fishing effort and bycatch in the near future.

## RESULTS

### Species caught in New Zealand fisheries

Thirty-one species were identified caught, these included 12 threatened species (IUCN 2008, table 1). Fourteen species were endemic to New Zealand. They were observed caught across a range of FMAs, except FMA8–9 where observer coverage was scant. Petrel species were caught predominantly in FMAs where they breed, while albatross were caught across a wider range of areas. Species for which over 100 individuals were returned during the six-year period were White-chinned Petrel, *Procellaria aequinoctialis* Linnaeus, 1758, White-capped Abatross, *Thalassarche steadi* (Falla, 1933), Sooty Shearwater, *Puffinus griseus* (J.F. Gmelin, 1789), Grey Petrel, *Procellaria cinerea* J.F. Gmelin, 1789, Salvin's Albatross, *Thalassarche cauta salvini* (Rothschild, 1893) and Buller's Albatross, *Thalassarche bulleri* (Rothschild, 1893). Examining the area-species relationships shows that albatross were caught mainly in southern areas (FMA3–7; table 2), and petrels spread across a wider range of areas (table 3). Both groups of seabirds were caught in all fishing methods. The number of albatross captured in TR fisheries was due to a preponderance of White-capped Albatross taken in FMA5–6. Similarly, petrel captures were mainly in FMA3–6 due to the influence of large numbers of White-chinned and Grey petrel captures in these areas in BLL and TR fisheries.

### Observer sampling of fishing effort

The percentage of fishing effort observed in the different fishing methods varied by vessel size, and was on average 4.6% across all fishing effort during the fishing years 1998–2004 (table 4). Observer coverage was over 10% of the total effort for large vessels across all methods, on average for the period of the study. Large SLL vessels had coverage over 50% in

**TABLE 2**
**Albatross species occurring in bycatch in New Zealand fisheries for the fishing years 1998–2004**

| Common name | Scientific name | IUCN threat status[2] | BLL[1] | | | | | | SLL | | | | | | TR | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | FMA3[3] | FMA4 | FMA5 | FMA6 | FMA7 | BLL Total | FMA1 | FMA2 | FMA3 | FMA5 | FMA7 | SLL Total | FMA2 | FMA3 | FMA4 | FMA5 | FMA6 | FMA7 | TR Total | Grand Total |
| White-capped Albatross | *Thalassarche steadi* | NT | | | | 1 | | 1 | | 1 | 2 | 33 | 5 | 41 | 4 | 25 | 4 | 423 | 274 | 27 | 757 | 799 |
| Salvin's Albatross | *Thalassarche salvini* | VU | 1 | 17 | 1 | 111 | | 130 | | 6 | | 1 | | 7 | 10 | 30 | 39 | 12 | 3 | | 94 | 231 |
| Buller's Albatross | *Thalassarche bulleri* | VU | | 2 | | | 1 | 3 | | 10 | 1 | 60 | 21 | 92 | | 18 | 17 | 30 | 6 | 12 | 83 | 178 |
| Antipodean Albatross | Diomedea antipodensis | VU | | | | | | | | 68 | | 2 | | 70 | | 1 | | | | | 1 | 71 |
| Campbell Albatross | *Thalassarche impavida* | VU | | | | | | | 2 | 30 | 1 | 2 | 1 | 36 | | | 1 | 1 | 1 | 7 | 10 | 46 |
| Black-browed Albatross | *Thalassarche melanophris* | EN | | | | 2 | | 2 | | 7 | | | | 7 | 1 | 1 | 2 | | 1 | 2 | 7 | 16 |
| Southern Royal Albatross | *Diomedea epomophora* | VU | | | | | | | | | 1 | 5 | | 6 | | 1 | 2 | 5 | | | 8 | 14 |
| Chatham Albatross | *Thalassarche eremita* | CR | | 7 | | | | 7 | | | | | | | | | 4 | | | | 4 | 11 |
| Light-mantled Albatross | *Phoebetria palpebrata* | NT | | | | | | | | | 1 | | 2 | 3 | | | | | | | | 3 |
| Northern Royal Albatross | *Diomedea sanfordi* | EN | | | | | | | | | | | | | | | 2 | | | | 2 | 2 |

[1] Bottom longline (BLL), surface longline (SLL) and trawl (TR) methods. [2] IUCN (2008).
[3] Fishing Management Area.

all years. For small vessels, coverage was very low, and lower than 2% in all except one method/year group. The proportion of fishing effort for which observer coverage was over 10%, as a result of the placement of observers on the large vessels was 45% for TR, and 8% for SLL and 25% for BLL. For some fishery strata, there was no fishing effort (observed or unobserved) corresponding to that method and vessel size, and these are shown as blanks in the relevant tables.

## Model diagnostics

For all methods, comparison of the observed number of seabird captures to that predicted by the model did not indicate any major systematic problems for either albatross or all seabirds combined. There is a tendency for the residuals to be positive which may indicate bycatch estimates are conservative (i.e., slightly underestimated), or this could be a result of the skewed distribution for the number of captures.

Approximately 6% of standardised residuals for TR and 5% for SLL were >3 which seemed reasonable (as the percentage would be 2.5% if they were normally distributed). The fraction of Year × Area × Season × Target Species × Vessel Class observations that lie below the 2.5$^{th}$ percentile of the distribution of the predicted observations is 2.7% and 4.8% lay above the 97.5$^{th}$ percentile for TR and 1.6% and 4.9% respectively for SLL.

For BLL, approximately 12% of standardised residuals are >3, which is high and may be due to insufficient variation in the predicted values or small sample size. There may have been insufficient variation in the predicted compared to the observed data (13.7% of observations were below the 2.5$^{th}$ percentile of the distribution of the predicted observations, 11.8% were above the 97.5$^{th}$ percentile) which may translate to the reported credible intervals being too narrow. However, this result may also be an artefact of only 51 Year × Area × Season × Target Species × Vessel Class combinations being observed (e.g., a small sample problem).

## Estimated seabird bycatch

For the 2004 year, estimated total seabird bycatch by area and vessel class is summarised in tables 5, 7 and 9 for TR, SLL and BLL methods, and for albatross bycatch for TR only (table 11). The rates of capture for corresponding groups of fishing effort are shown (tables 6, 8, 10 and 12 respectively). Total estimated seabird bycatch for all years of the study is shown in figures 2, 3, 4 respectively for TR, SLL, BLL, and in figure 5 for albatross in TR fisheries only.

## Trawl fisheries

Total seabird bycatch by large TR vessels was estimated to have been increasing during the 1998–2004 fishing years, and was between 2000–3000 birds for most of this period (width of 95% CI's ~800 birds; fig. 2), with bycatch in 1998 estimated to be slightly lower, and that in 2001 estimated to be higher. It has remained stable, at around 2500 birds from 2002 to 2004. For small TR vessels, total seabird bycatch was estimated to be relatively low (<200), but with a high degree of uncertainty (width of 95% CI's ~250 birds; fig. 2). The widths of the confidence intervals for small vessels indicate that these estimates are relatively poorly known. Most of the seabird bycatch is predicted to occur in the Hoki, *Macruronus novaezealandiae* (Hector, 1871), and squid fisheries (MacKenzie& Fletcher 2005).

**TABLE 3**
**Petrel species occurring in bycatch in New Zealand fisheries for the fishing years 1998–2004**

| Common name | Scientific name | IUCN threat status[3] | BLL[1] | | | | | | | SLL | | | | | | | TR | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | FMA1[2] | FMA2 | FMA3 | FMA4 | FMA5 | FMA6 | BLL Total | FMA1 | FMA2 | FMA3 | FMA5 | FMA7 | FMA10 | SLL Total | FMA1 | FMA2 | FMA3 | FMA4 | FMA5 | FMA6 | FMA7 | TR Total | Grand Total |
| White-chinned Petrel | *Procellaria aequinoctialis* | VU | | | 6 | 338 | 222 | 178 | 744 | 1 | 5 | 7 | 10 | | | 23 | | | 13 | 19 | 97 | 40 | | 169 | 936 |
| Sooty Shearwater | *Puffinus griseus* | NT | | 2 | 1 | 1 | 88 | 2 | 94 | | 6 | | | | | 6 | | 2 | 256 | 28 | 238 | 57 | | 581 | 681 |
| Grey Petrel | *Procellaria cinerea* | NT | 11 | 1 | 6 | 32 | | 322 | 372 | 2 | 89 | | | | | 91 | | | 1 | 3 | | 4 | | 8 | 471 |
| Flesh-footed Shearwater | *Puffinus carneipes* | LC | 16 | | | | | | 16 | | 14 | | | | | 14 | 3 | 2 | | | | | | 5 | 35 |
| Short-tailed Shearwater | *Puffinus tenuirostris* | LC | | | | | | | | | | | | | | | | | | | 33 | | | 33 | 33 |
| Cape Pigeon | *Daption capense* | LC | | 1 | | 7 | 3 | 5 | 16 | | | | | | | | | 1 | | 1 | 4 | 2 | 5 | 13 | 29 |
| Grey-faced Petrel | *Pterodroma macroptera* | LC | | | | | | | | 4 | 3 | | | | 10 | 17 | | | | 1 | | | | 1 | 18 |
| Northern Giant Petrel | *Macronectes halli* | NT | | | | | | 3 | 3 | | 1 | | | | | 1 | | | 1 | 7 | | | | 8 | 12 |
| Black Petrel | *Procellaria parkinsoni* | VU | 2 | | | | | | 2 | 7 | | | | | | 7 | | | | | | | | | 9 |
| Common Diving -Petrel | *Pelecanoides urinatrix* | LC | | | | | 3 | 1 | 4 | | | | | | | | | | | | 2 | 2 | | 4 | 8 |
| Southern Giant Petrel | *Macronectes giganteus* | NT | | | | | | 4 | 4 | | 2 | | | | | 2 | | | | | | | | | 6 |
| FairyPrion | *Pachyptila turtur* | LC | | | | | | | | | | | | | | | | | | | | | 6 | 6 | 6 |
| Westland Petrel | *Procellaria westlandica* | VU | | | | | | | | | 1 | | 1 | 1 | | 3 | | | | | | | 1 | 1 | 4 |
| White-headed Petrel | *Pterodroma lessonii* | LC | | | | | | | | | | | | | 2 | 2 | | | | | | | | | 2 |
| Buller's Shearwater | *Puffinus bulleri* | NT | 2 | | | | | | 2 | | | | | | | | | | | | | | | | 2 |
| Fluttering Shearwater | *Puffinus gavia* | LC | 2 | | | | | | 2 | | | | | | | | | | | | | | | | 2 |
| Grey-backed Storm-Petrel | *Garrodia nereis* | LC | | | | | | | | | | | | | | | | | | | | 2 | | 2 | 2 |
| Antarctic Prion | *Pachyptila desolata* | LC | | | | | | | | | | | | | | | | | | 1 | | 1 | | 2 | 2 |
| White-faced Storm-Petrel | *Pelagodroma marina* | LC | | | | 2 | | | 2 | | | | | | | | | | | | | | | | 2 |
| Black-bellied Storm -Petrel | *Fregetta tropica* | LC | | | | | | | | | | | | | | | | | | | 1 | | | 1 | 1 |
| Broad-billed Prion | *Pachyptila vittata* | LC | | | | | 1 | | 1 | | | | | | | | | | | | | | | | 1 |

[1] Bottom longline (BLL), surface longline (SLL) and trawl (TR) methods. [2] Fishery Management Area.
[3] IUCN (2008).

**TABLE 4**
**Fishing events and observed seabird bycatch for New Zealand fisheries, 1998–2004 fishing years**

| Fishing Year | Fishing method[1] | Vessel size (m) | Total events[4] | % events observed | Total observed seabird (and albatross) bycatch |
|---|---|---|---|---|---|
| 1998 | TR | <28 | 115481 | 0 | 10 (4)[2] |
| 1999 | TR | <28 | 103047 | 1 | 17 (7) |
| 2000 | TR | <28 | 86617 | 1 | 8 (8) |
| 2001 | TR | <28 | 83571 | 0 | 11 (10) |
| 2002 | TR | <28 | 79952 | 1 | 5 (5) |
| 2003 | TR | <28 | 80053 | 0 | 2 (2) |
| 2004 | TR | <28 | 76065 | 0 | 4 (4) |
| 1999 | TR | >28 | 80428 | 9 | 269 (128) |
| 2000 | TR | >28 | 71519 | 10 | 173 (102) |
| 2001 | TR | >28 | 69376 | 13 | 683 (244) |
| 2002 | TR | >28 | 75544 | 10 | 322 (189) |
| 2003 | TR | >28 | 72986 | 9 | 298 (126) |
| 2004 | TR | >28 | 66097 | 10 | 290 (167) |
| 1998 | SLL | <28 | 3119 | 3 | 45 |
| 1999 | SLL | <28 | 5307 | 1 | 10 |
| 2000 | SLL | <28 | 6658 | 1 | 34 |
| 2001 | SLL | <28 | 7631 | 2 | 38 |
| 2002 | SLL | <28 | 8336 | 1 | 87 |
| 2003 | SLL | <28 | 7242 | 0 | –[3] |
| 2004 | SLL | <28 | 5007 | 2 | 2 |
| 1998 | SLL | >28 | 609 | 59 | 171 |
| 1999 | SLL | >28 | 635 | 65 | 74 |
| 2000 | SLL | >28 | 380 | 70 | 40 |
| 2001 | SLL | >28 | 533 | 51 | 15 |
| 2002 | SLL | >28 | 488 | 56 | 80 |
| 2003 | SLL | >28 | 663 | 91 | 115 |
| 2004 | SLL | >28 | 571 | 82 | 70 |
| 1999 | BLL | <28 | 22796 | 0 | 0 |
| 2000 | BLL | <28 | 23214 | 0 | – |
| 2001 | BLL | <28 | 22709 | 0 | 26 |
| 2002 | BLL | <28 | 20376 | 0 | – |
| 2003 | BLL | <28 | 18979 | 0 | 1 |
| 2004 | BLL | <28 | 17342 | 1 | 10 |
| 1999 | BLL | >28 | 10456 | 7 | 93 |
| 2000 | BLL | >28 | 9034 | 11 | 203 |
| 2001 | BLL | >28 | 7978 | 11 | 509 |
| 2002 | BLL | >28 | 5387 | 20 | 431 |
| 2003 | BLL | >28 | 4278 | 37 | 426 |
| 2004 | BLL | >28 | 6146 | 12 | 122 |
| 1998 | TR | >28 | 80626 | 8 | 96 (37) |
| Total effort | All | All | 1357236 | 4.6 | 4790 (1033) |

[1] Bottom longline (BLL), surface longline (SLL) and trawl (TR) methods.
[2] Bracketed numbers indicate the number of albatrosses observed for trawl vessels.
[3] – indicates no effort was observed, thus observed seabird bycatch cannot be reported.
[4] Fishing events are tows for trawling, and sets for longlining.
Numbers of birds here are not the same as those identified to species level in tables 1–3, as they include unidentified seabirds.

**TABLE 5**
**Estimated seabird bycatch in New Zealand trawl fisheries in the 2004 fishing year**

| Vessel length (m) | FMA[1] | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 9 | Other |
| >28 | 0 | 25 | 480 | 215 | 702 | 970 | 164 | 7 | 14 |
| | (0, 48)[2] | (4, 67) | (379, 601) | (157, 286) | (589, 831) | (821, 1149) | (125, 214) | (2, 21) | (2, 45) |
| <28 | 0 | 3 | 39 | 1 | 6 | 2 | 15 | 0 | 0 |
| | (0, 23) | (0, 22) | (17, 134) | (1, 3) | (1, 24) | (0, 8) | (5, 62) | (0, 3) | (0, 2) |

[1] Fishery Management Area.
[2] Figures in brackets indicate the 95% credibility interval for each cell above.

**TABLE 6**
**Estimated seabird bycatch rates in New Zealand trawl fisheries per 100 tows in the 2004 fishing year**

| Vessel length (m) | FMA[1] | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 9 | Other |
| >28 | 0.00 | 0.88 | 6.36 | 2.43 | 11.44 | 5.63 | 2.01 | 0.29 | 0.12 |
| | (0.00, 4.11)[2] | (0.14, 2.35) | (5.02, 7.96) | (1.77, 3.23) | (9.60, 13.54) | (4.76, 6.66) | (1.54, 2.63) | (0.08, 0.88) | (0.02, 0.38) |
| <28 | 0.00 | 0.03 | 0.17 | 0.24 | 0.17 | 0.19 | 0.08 | 0.00 | 0.00 |
| | (0.00, 0.20) | (0.00, 0.20) | (0.07, 0.59) | (0.24, 0.73) | (0.03, 0.67) | (0.00, 0.75) | (0.03, 0.33) | (0.00, 0.09) | (0.00, 0.06) |

[1] Fishery Management Area.
[2] Figures in brackets indicate the 95% credibility interval for each cell above.

**TABLE 7**
**Estimated seabird bycatch in New Zealand surface longline fisheries in the 2004 fishing year**

| Vessel length (m) | FMA[1] | | | | | | |
|---|---|---|---|---|---|---|---|
| | 1 | 2 and 4 | 3 and 6 | 5 | 7 | 8 and 9 | Other |
| >28 | 0 | 164 | 3 | 46 | 18 | 0 | 0 |
| | (0, 0)[2] | (46, 590) | (3,3) | (46,49) | (16, 29) | (0, 0) | (0, 3) |
| <28 | 678 | 999 | 0 | 1 | 2 | 0 | 2 |
| | (139, 1,602) | (193, 2,205) | (0, 2) | (0,6) | (0, 11) | (0, 1) | (1, 8) |

[1] Fishery Management Area.
[2] Figures in brackets indicate the 95% credibility interval for each cell above.

**TABLE 8**
**Estimated seabird bycatch rates in New Zealand surface longline fisheries per set in the 2004 fishing year**

| Vessel length (m) | FMA[1] | | | | | | |
|---|---|---|---|---|---|---|---|
| | 1 | 2 and 4 | 3 and 6 | 5 | 7 | 8 and 9 | Other |
| >28 | 0.00 | 1.89 | 1.00 | 0.20 | 0.08 | 0.00 | 0.00 |
| | (0.00, 0.00)[2] | (0.53, 6.78) | (1.00, 1.00) | (0.20, 0.21) | (0.07,0.13) | (0.00,0.00) | (0.00, 0.60) |
| <28 | 0.44 | 0.40 | 0.00 | 0.03 | 0.01 | 0.00 | 0.07 |
| | (0.09, 1.03) | (0.08, 0.88) | (0.00, 0.15) | (0.00, 0.18) | (0.00,0.03) | (0.00, 0.00) | (0.03, 0.28) |

[1] Fishery Management Area.
[2] Figures in brackets indicate the 95% credibility interval for each cell above.

**TABLE 9**
**Estimated seabird bycatch in New Zealand bottom longline fisheries in the 2004 fishing year**

| Vessel length (m) | FMA[1] | | | | |
|---|---|---|---|---|---|
| | 1, 2, 8, 9 and 10 | 3 and 4 | 5 and 7 | 6 | Other |
| >28 | 1 | 186 | 104 | 341 | 0 |
| | (0, 6)[2] | (104, 393) | (89, 126) | (251, 465) | (0, 1) |
| <28 | 247 | 14 | 64 | 0 | 0 |
| | (49, 1,671) | (1, 140) | (8, 530) | (0, 3) | (0, 0) |

[1] Fishery Management Area.
[2] Figures in brackets indicate the 95% credibility interval for each cell above.

**TABLE 10**
**Estimated seabird bycatch rates in New Zealand bottom longline fisheries per set in the 2004 fishing year**

| Vessel length (m) | FMA[1] | | | | |
|---|---|---|---|---|---|
| | 1, 2, 8, 9 and 10 | 3 and 4 | 5 and 7 | 6 | Other |
| >28 | 0.00 | 0.05 | 0.23 | 0.11 | 0.00 |
| | (0.01, 0.04) | (0.09, 0.19) | (0.27, 0.32) | (0.15, 0.21) | (0.00, 0.00) |
| <28 | 0.00 | 0.00 | 0.01 | 0.00 | 0.00 |
| | (0.02, 0.11) | (0.02, 0.21) | (0.05, 0.38) | (0.00, 0.75) | (0.00, 0.00) |

[1] Fishery Management Area.
[2] Figures in brackets indicate the 95% credibility interval for each cell above.

**TABLE 11**
**Estimated albatross bycatch in New Zealand trawl fisheries in the 2004 fishing year**

| Vessel length (m) | FMA[1] | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 9 | Other |
| >28 | 0 | 17 | 128 | 112 | 368 | 585 | 82 | 0 | 2 |
| | (0, 5)[2] | (6, 36) | (90, 177) | (74, 163) | (300, 448) | (482, 708) | (57, 115) | (0, 7) | (0, 21) |
| <28 | 1 | 19 | 81 | 2 | 18 | 7 | 62 | 0 | 0 |
| | (0, 17) | (2, 77) | (18, 349) | (1, 8) | (3, 81) | (1, 27) | (12, 300) | (0, 6) | (0, 6) |

[1] Fishery Management Area.
[2] Figures in brackets indicate the 95% credibility interval for each cell above.

**Table 12**
**Estimated albatross bycatch rates in New Zealand trawl fisheries per 100 tows in the 2004 fishing year**

| Vessel length (m) | FMA[1] | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 9 | Other |
| >28 | 0 | 0.6 | 1.7 | 1.27 | 6 | 3.39 | 1.01 | 0 | 0.02 |
| | (0.00, 0.43)[2] | (0.21, 1.26) | (1.19, 2.34) | (0.84, 1.84) | (4.89, 7.30) | (2.80, 4.11) | (0.70, 1.41) | (0.00, 0.29) | (0.00, 0.18) |
| <28 | 0.01 | 0.17 | 0.36 | 0.49 | 0.5 | 0.66 | 0.33 | 0 | 0 |
| | (0.00, 0.15) | (0.02, 0.70) | (0.08, 1.54) | (0.24, 1.95) | (0.08, 2.25) | (0.09, 2.55) | (0.06, 1.58) | (0.00, 0.19) | (0.00, 0.17) |

[1] Fishery Management Area.
[2] Figures in brackets indicate the 95% credibility interval for each cell above.

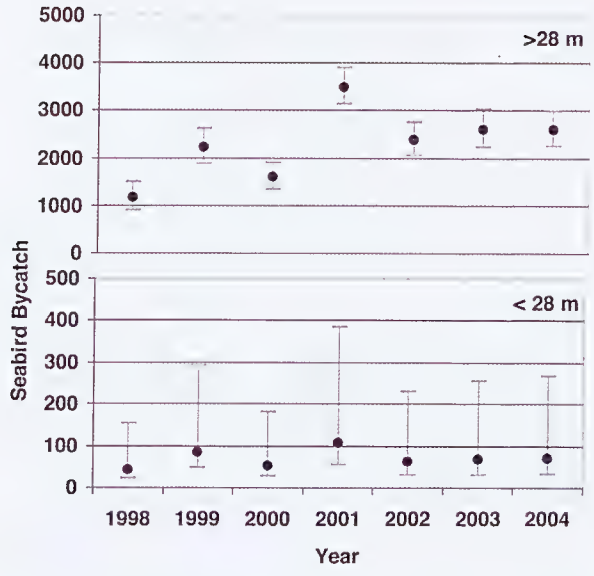


*FIG. 2 — Estimated total seabird bycatch in New Zealand trawl fisheries for the fishing years 1998–2004 for vessels greater than 28 m in length (upper graph) and less than 28 m (lower graph).*

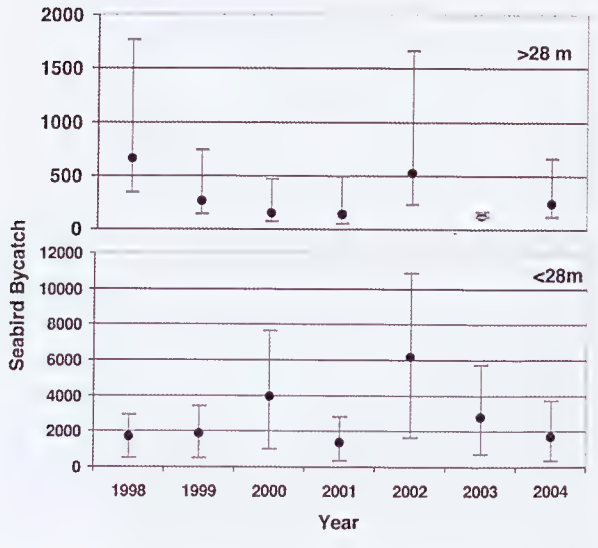


*FIG. 3 — Estimated total seabird bycatch in New Zealand surface longline fisheries for the fishing years 1998–2004 for vessels greater than 28 m in length (upper graph) and less than 28 m (lower graph).*

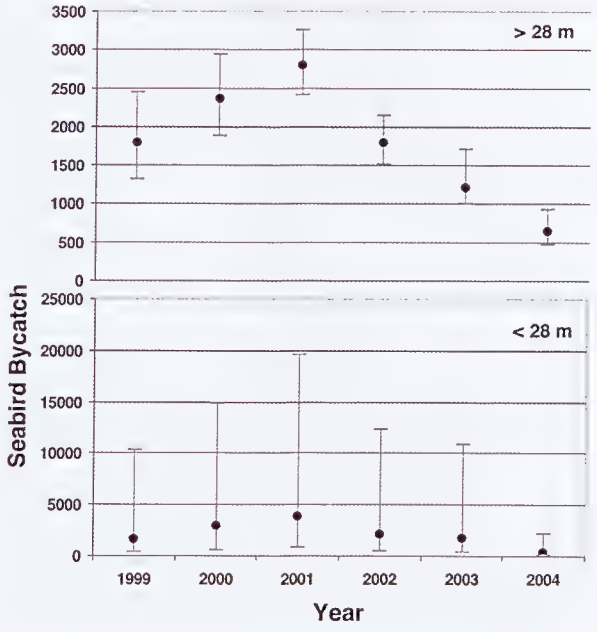


*FIG. 4 — Estimated total seabird bycatch in New Zealand bottom longline fisheries for the fishing years 1999–2004 for vessels greater than 28 m in length (upper graph) and less than 28 m (lower graph).*

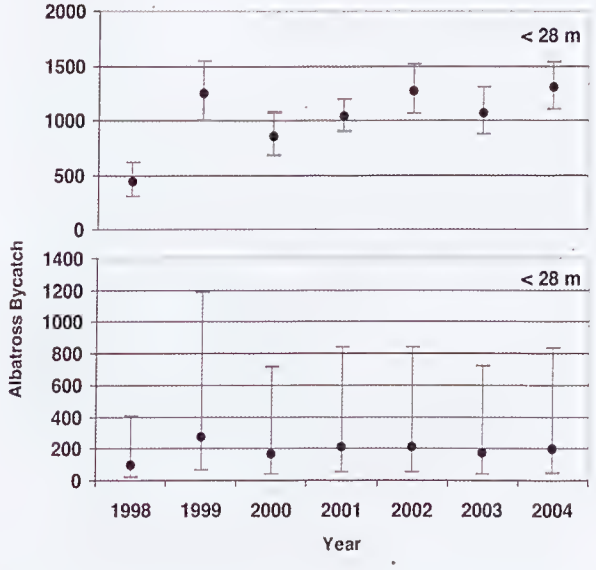


*FIG. 5 — Estimated albatross bycatch in New Zealand trawl fisheries for the fishing years 1998–2004 for vessels greater than 28 m in length (upper graph) and less than 28 m (lower graph).*

Capture rates varied by area (table 6) and were highest in FMA3–7 where rates were between 2–11 birds per 100 tows in 2004. There was variation in the rates of capture between years, but since 2002, these had been relatively stable and at similar rates to those in 2004 (MacKenzie & Fletcher 2005).

## Surface longline fisheries

Total seabird bycatch by large SLL vessels was generally estimated to be less than 500 birds (width of 95% CI's ~500 birds in most years), although the upper bound of the credible intervals in the 1998 and 2002 fishing years was relatively high at approximately 1700 birds (fig. 3). In 2004, large vessels

were estimated to catch approximately 200 birds (table 7). For small vessels, the 95% credible intervals for total seabird bycatch generally ranged between 500 and 4000 birds, fig. 3), with notable exceptions being those for the 2000 and 2002 fishing years for total bycatch when CIs were larger. Most of the seabird bycatch for SLL fishing was predicted to occur in the Big-eye Tuna, *Thunnus obesus* (Lowe, 1839), fishery (MacKenzie & Fletcher 2005).

Catch rates (birds per set) for large SLL vessels in 2004 were 1.9 birds per set in FMA2 and 4, 1.0 birds per set in FMA3 and 6, with lower rates elsewhere (table 8). On average through the study period, rates were around or less than one bird per set (MacKenzie & Fletcher 2005). For small vessels catch rates showed no particular pattern through time, and had average rates of around 0.4 birds per set in 2003 and 2004 (MacKenzie & Fletcher 2005).

## Bottom longline fisheries

Total seabird bycatch for large BLL vessels was estimated to have increased by approximately 500 birds per year from 1999–2001, but was estimated to have steadily decreased in recent years to approximately 600 seabirds (95% CI 500–1,000) in 2004 (table 9, fig. 4, but this apparent increase prior to 2001 may be due to problems with reporting of data). Seabird bycatch by smaller vessels in bottom longline fisheries was estimated with much less precision, with point estimates generally in the range of approximately 2000–4000 birds, but the width of the 95% credible intervals was larger than 10 000 seabirds in five of the six years (fig. 4) indicating a very high degree of uncertainty. Most of the seabird bycatch for large vessels is predicted to occur on sets targeting Ling, *Genypterus blacodes* Block & Schneider 1801, and on sets targeting snapper, *Pagrus auratus* (Block & Schneider, 1801), for small vessels (MacKenzie & Fletcher 2005).

Capture rates for BLL fisheries were highest in FMA3–7 (table 10), and varied between 0.05 and 0.23 birds per set. Catch rates varied through time, in a similar way to capture totals and these differences may reflect problems with observer data reporting in the fishery. These problems were resolved by 2003, and rates in 2003 were around 0.3 birds per set, decreasing to 0.1 birds per set in 2004 on average across areas (MacKenzie & Fletcher 2005). Rates of capture were estimated to be consistently low for small BLL vessels (table 10, MacKenzie & Fletcher 2005).

## Albatross bycatch in trawl fisheries

Albatross bycatch by large TR vessels was estimated to be approximately 450 birds (95% credible interval 300–600 birds) in the 1998 fishing year, and between 900 and 1500 in subsequent years (width of 95% CI's ~500 birds; fig. 5). Approximately 1300 albatross were estimated killed on large trawl vessels in 2004 (table 11). For small TR vessels, albatross bycatch tended to be less than 200 birds, but there is a great deal of uncertainty surrounding these estimates (width of 95% CI's ~800 birds; fig. 5). Most of the bycatch is predicted to occur in Hoki and squid fisheries (MacKenzie & Fletcher 2005).

Catch rates of albatross for large TR vessels were highest in FMA5 (table 12), at around six birds per 100 tows in 2004. The average rate for 2004 (near to two birds per 100 tows) was higher than that for large vessels on average for the proceeding five years, when it was close to 1.5 birds per 100 tows (MacKenzie & Fletcher 2005). For small TR vessels, catch rates in FMA3–7 were highest (table 12), but had been stable through time at around 0.2 birds per 100 tows on average (MacKenzie & Fletcher 2005).

## Assessing relative risk of seabird capture

Using the outputs of the predictive models developed for estimation of seabird bycatch in each area, season and fishing method, we explored relative importance of bycatch for different fishery areas and methods. Results are tabulated to enable detailed examination of the results (tables 13–16), and represented graphically to present an overall perspective of the relative bycatch and associated error (figs 6–9). During modelling, data for some FMAs were combined to reduce the uncertainty associated with total seabird capture estimates.

The estimates of total bycatch (number of birds) across all vessel sizes, methods, areas and seasons, for both 2004 and the average over 1998–2004 were calculated (table 13). The results for the 2004 fishing year showed similar rankings to those for the six-year average for both datasets. We therefore discuss only the 2004 results for simplicity, ranked in descending order of the estimated bycatch in 2004. For ease of presentation, we have removed those fishery strata for which the estimate in 2004 was less than 0.5% of the total. The estimated seabird bycatch reported included in table 13 comprised 99% of the total estimated bycatch for the 2004 fishing year. Each estimate is also shown as a percentage of the sum of the estimates across all fishery strata (% total in table 13). This indicates where the greatest "risk" of capturing seabirds lies, as estimated from the 2004 predicted capture estimates by vessel size, area, season and method. The uncertainties associated with these estimates are shown as a percentage of the total estimated bycatch for 2004 (table 14).

The analysis indicates that around 63% of total seabird bycatch occurs on large vessels and around 36% on small vessels.

Bycatch was estimated to have been highest in FMA6 (25%), followed by FMA1 and FMA5 (13% each), and FMA3 (9%). Combined estimates for FMA1, 2, 8 and 10 and for FMA3–4 may add up to an additional 4% to the totals for FMA1 and 3. Bycatch was highest in the spring and summer (34% each) and lowest in the spring and winter (19% and 12% respectively). The most important uncertainties were associated with small longline vessels, especially SLL vessels in FMA2 and 4 and BLL vessels in FMA1–2 and FMA8–10 (table 14).

For TR fisheries, which accounted for 46% of the total bycatch (table 13), highest total bycatch occurred in spring and summer across all areas, but particularly in FMA3, FMA5 and FMA6 for large vessels (39% of total captures) (fig. 6).

For SLL fisheries, which accounted for 35% of the total bycatch, the highest bycatch was estimated for FMA2 and 4 in summer and autumn for small vessels (16% of total bycatch), and FMA1, across all seasons for small vessels (13% of total bycatch) (fig. 7). There was high uncertainty associated with these estimates. Large vessels contributed a relatively small proportion of total SLL bycatch (4% of total bycatch), mainly in FMA2 and FMA4 in the summer.

BLL fisheries accounted for around 18% of the total bycatch, and spring showed relatively high bycatch totals across all areas for large vessels (6% of the total bycatch), while most bycatch was recorded in FMA3, FMA4 and FMA6 (3% of the total bycatch). For small vessels, very

**TABLE 13**
**Estimated bycatch of seabirds in New Zealand fisheries for the 2004 fishing year and the average over the 1998–2004 fishing years**

| Vessel size (m)[1] | Fishing method[2] | FMA[3] | Season[4] | 2004 | | 1998–2004[6] | |
|---|---|---|---|---|---|---|---|
| | | | | Birds[5] | % Total birds | Birds | % Total birds |
| Small | SLL | 2, 4 | Summer | 519 | 10 | 659 | 7 |
| Large | TR | 6 | Autumn | 377 | 7 | 230 | 2 |
| Large | TR | 6 | Summer | 372 | 7 | 223 | 2 |
| Large | TR | 5 | Autumn | 338 | 6 | 241 | 3 |
| Small | SLL | 2, 4 | Autumn | 320 | 6 | 446 | 5 |
| Large | TR | 5 | Summer | 298 | 5 | 309 | 3 |
| Small | SLL | 1 | Winter | 248 | 5 | 518 | 6 |
| Large | TR | 3 | Autumn | 225 | 4 | 332 | 4 |
| Large | TR | 6 | Spring | 202 | 4 | 120 | 1 |
| Small | SLL | 1 | Spring | 171 | 3 | 396 | 4 |
| Small | SLL | 1 | Summer | 154 | 3 | 297 | 3 |
| Large | BLL | 6 | Spring | 151 | 3 | 333 | 4 |
| Large | SLL | 2, 4 | Summer | 141 | 3 | 45 | 0 |
| Large | TR | 3 | Summer | 133 | 2 | 169 | 2 |
| Large | TR | 7 | Winter | 108 | 2 | 92 | 1 |
| Large | TR | 3 | Spring | 93 | 2 | 109 | 1 |
| Large | BLL | 5, 7 | Spring | 89 | 2 | 191 | 2 |
| Large | BLL | 6 | Autumn | 88 | 2 | 278 | 3 |
| Large | TR | 4 | Autumn | 87 | 2 | 77 | 1 |
| Small | BLL | 1, 2, 8–10 | Autumn | 87 | 2 | 342 | 4 |
| Small | SLL | 1 | Autumn | 82 | 2 | 207 | 2 |
| Small | BLL | 1, 2, 8–10 | Summer | 81 | 1 | 293 | 3 |
| Small | SLL | 2, 4 | Spring | 77 | 1 | 95 | 1 |
| Large | BLL | 6 | Summer | 66 | 1 | 245 | 3 |
| Small | BLL | 1, 2, 8–10 | Winter | 64 | 1 | 294 | 3 |
| Large | TR | 4 | Summer | 63 | 1 | 65 | 1 |
| Large | TR | 5 | Spring | 57 | 1 | 76 | 1 |
| Small | SLL | 2, 4 | Winter | 56 | 1 | 38 | 0 |
| Large | TR | 4 | Spring | 54 | 1 | 56 | 1 |
| Large | BLL | 3, 4 | Spring | 54 | 1 | 260 | 3 |
| Large | BLL | 3, 4 | Winter | 54 | 1 | 191 | 2 |
| Large | SLL | 5 | Autumn | 46 | 1 | 43 | 0 |
| Large | BLL | 3, 4 | Autumn | 44 | 1 | 65 | 1 |
| Large | TR | 7 | Autumn | 42 | 1 | 52 | 1 |
| Large | BLL | 6 | Winter | 33 | 1 | 132 | 1 |
| Large | BLL | 3, 4 | Summer | 31 | 1 | 39 | 0 |
| Small | BLL | 5, 7 | Spring | 29 | 1 | 208 | 2 |
| Large | TR | 3 | Winter | 28 | 1 | 22 | 0 |

[1] Small = < 28 m; Large = >28 m.
[2] Bottom longline (BLL), surface longline (SLL) and trawl (TR) methods.
[3] Fishery Management Area.
[4] Spring = Oct–Dec; Summer = Jan–Mar; Autumn = Apr–Jun; Winter = Jul–Sep
[5] Vessel-method-area-season combinations with less than 0.5 of the estimated bycatch in the 2004 fishing year have been omitted from the table.
[6] 1999–2004 for bottom longline.

**TABLE 14**
**Uncertainty (width of 95 credible interval) associated with estimates of bycatch of seabirds New Zealand fisheries**

| Vessel size[1] | Fishing method[2] | FMA[3] | Season[4] | Birds | % total birds[5] |
|---|---|---|---|---|---|
| Small | SLL | 2, 4 | Summer | 991 | 18 |
| Small | SLL | 2, 4 | Autumn | 803 | 15 |
| Small | SLL | 1 | Winter | 764 | 14 |
| Small | BLL | 1, 2, 8–10 | Summer | 564 | 10 |
| Small | BLL | 1, 2, 8–10 | Autumn | 559 | 10 |
| Large | SLL | 2, 4 | Summer | 521 | 10 |
| Small | BLL | 1, 2, 8–10 | Winter | 475 | 9 |
| Small | SLL | 1 | Spring | 378 | 7 |
| Small | SLL | 1 | Summer | 268 | 5 |
| Small | BLL | 5, 7 | Spring | 260 | 5 |
| Small | SLL | 2, 4 | Spring | 210 | 4 |
| Small | SLL | 1 | Autumn | 205 | 4 |
| Small | SLL | 2, 4 | Winter | 197 | 4 |
| Large | TR | 6 | Autumn | 176 | 3 |
| Small | BLL | 5, 7 | Summer | 169 | 3 |
| Large | TR | 5 | Autumn | 155 | 3 |
| Large | BLL | 6 | Spring | 133 | 2 |
| Large | TR | 3 | Autumn | 125 | 2 |
| Large | TR | 6 | Spring | 123 | 2 |
| Large | TR | 6 | Summer | 119 | 2 |
| Large | BLL | 3, 4 | Autumn | 115 | 2 |
| Large | TR | 5 | Summer | 111 | 2 |

[1] Small = < 28 m; Large = >28 m.
[2] Bottom longline (BLL), surface longline (SLL) and trawl (TR) methods.
[3] Fishery Management Area.
[4] Spring = Oct–Dec; Summer = Jan–Mar; Autumn = Apr–Jun; Winter = Jul–Sep.
[5] The uncertainty is also shown as a percentage of the total estimated bycatch for the 2004 fishing year. The uncertainties have been ranked in descending order and those vessel-method-area-season combinations for which the uncertainty was less than 2% of the estimated total bycatch in the 2004 fishing year have been omitted from the table.

high uncertainty was associated with all estimates with an estimated 5 % of total bycatch from this group, and little confidence can be placed in these results.

The bycatch estimates for albatross in TR fisheries and uncertainties in the 2004 fishing year are shown in tables 15 and 16. The fishery strata contained in these tables comprise 94% of the total estimated albatross bycatch for the period, and estimates of less than 2% of the albatross bycatch are not discussed. Large vessels accounted for around 83% of the estimated albatross mortality, and small vessels, around 11%. Most albatross were estimated caught in FMA6 (39%), followed by FMA5 (24%) and FMA3 (15%). Albatross mortalities in these three areas comprised around 78% of the total albatross mortality. Most albatross were caught in the summer (38%) and autumn (32%), followed by spring (17%) and winter (7%).

**TABLE 15**
**Estimated bycatch of albatrosses in New Zealand trawl fisheries, for the 2004 fishing year and the average over the 1998–2004 fishing years**

| Vessel size[1] | FMA[2] | Season[3] | 2004 | | 1998-2004[5] | |
|---|---|---|---|---|---|---|
| | | | Birds | % total birds[4] | Birds | % total birds |
| Large | 6 | Summer | 270 | 18 | 154 | 13 |
| Large | 5 | Summer | 186 | 12 | 198 | 16 |
| Large | 6 | Autumn | 183 | 12 | 91 | 7 |
| Large | 5 | Autumn | 153 | 10 | 84 | 7 |
| Large | 6 | Spring | 122 | 8 | 65 | 5 |
| Large | 7 | Winter | 55 | 4 | 49 | 4 |
| Large | 3 | Summer | 48 | 3 | 54 | 4 |
| Large | 3 | Autumn | 44 | 3 | 73 | 6 |
| Large | 4 | Spring | 36 | 2 | 36 | 3 |
| Large | 4 | Summer | 35 | 2 | 35 | 3 |
| Large | 4 | Autumn | 33 | 2 | 29 | 2 |
| Large | 3 | Spring | 28 | 2 | 33 | 3 |
| Small | 3 | Autumn | 28 | 2 | 25 | 2 |
| Small | 7 | Autumn | 26 | 2 | 20 | 2 |
| Small | 3 | Summer | 24 | 2 | 26 | 2 |
| Large | 5 | Spring | 23 | 2 | 34 | 3 |
| Small | 3 | Spring | 23 | 2 | 20 | 2 |
| Large | 7 | Autumn | 17 | 1 | 20 | 2 |
| Small | 7 | Spring | 15 | 1 | 11 | 1 |
| Small | 7 | Winter | 12 | 1 | 8 | 1 |
| Small | 7 | Summer | 9 | 1 | 9 | 1 |
| Large | 3 | Winter | 8 | 1 | 7 | 1 |
| Large | 6 | Winter | 8 | 1 | 8 | 1 |

[1] Small = < 28 m; Large = >28 m.
[2] Fishery Management Area.
[3] Spring = Oct–Dec; Summer = Jan–Mar; Autumn = Apr–Jun; Winter = Jul–Sep.
[4] Vessel-area-season combinations with less than 0.5% of the estimated bycatch in the 2004 fishing year have been omitted from the table.
[5] 1999–2004 for bottom longline.

## DISCUSSION

### Modelling approach

The approaches used here are novel for New Zealand fisheries for estimation of seabird bycatch, as previously ratio estimators were used to generate estimates for specific target fisheries (Baird 2004a, b, c, 2005, Baird & Smith 2007). Model-based estimation has been used recently, but only for specific target fisheries (Baird & Smith 2007). We approached the problem of estimation from the desire to provide an overall assessment of the importance of bycatch in different sectors of the longline and trawl fishery (e.g., by vessel size, method and area). The modelling approach used was appropriate for this analysis, allowing comparison of likelihood of seabird bycatch across broad areas of the fishery, including areas where observer data were sparse, with corresponding uncertainty estimates to guide the interpretation of the results.

**TABLE 16**
**Uncertainty (width of 95 credible interval) associated with estimates of bycatch of albatross in New Zealand trawl fisheries in the 2004 fishing year**

| Vessel size[1] | FMA[2] | Season[3] | Birds | % total[4] |
|---|---|---|---|---|
| Small | 7 | Autumn | 124 | 8 |
| Small | 3 | Autumn | 121 | 8 |
| Large | 6 | Autumn | 104 | 7 |
| Small | 3 | Summer | 104 | 7 |
| Large | 6 | Summer | 101 | 7 |
| Large | 6 | Spring | 90 | 6 |
| Large | 5 | Summer | 86 | 6 |
| Small | 3 | Spring | 86 | 6 |
| Large | 5 | Autumn | 81 | 6 |
| Small | 7 | Spring | 77 | 5 |
| Small | 7 | Winter | 53 | 4 |
| Small | 7 | Summer | 45 | 3 |
| Large | 3 | Summer | 43 | 3 |
| Large | 4 | Spring | 40 | 3 |
| Large | 3 | Autumn | 39 | 3 |
| Large | 7 | Winter | 39 | 3 |
| Large | 4 | Summer | 37 | 3 |
| Large | 4 | Autumn | 36 | 2 |
| Large | 3 | Spring | 30 | 2 |

[1] Small = < 28 m; Large = >28 m.
[2] Fishery Management Area.
[3] Spring = Oct–Dec; Summer = Jan–Mar; Autumn = Apr–Jun; Winter = Jul–Sep.
[4] The uncertainty is also shown as a percentage of the total estimated bycatch for 2004. The uncertainties have been ranked in descending order and those vessel-area-season combinations for which the uncertainty was less than 2% of the estimated total bycatch in the 2004 fishing year have been omitted from the table.

We consider the important outstanding problems in the analysis were:
(a) The spread of observer coverage across vessels. This was the single most important problem associated with providing improved estimates of seabird bycatch. Increasing the representativeness of sampling across the fishing effort, seasons and particularly vessel sizes would greatly improve the accuracy of bycatch estimation. Given the manner in which observers have been allocated to vessels in the past, and the very low levels of observer coverage of small vessels, caution should be exercised when interpreting the estimated level of bycatch from currently collected scientific observer data regardless of the statistical method employed to provided that estimate. At present it may only be possible to reliably estimate seabird bycatch for a small number of fisheries. This considerably reduces the utility of the results of the study to understanding the seabird bycatch problem across New Zealand fisheries at a global level and from the perspective of individual populations. The relative importance of different components of the total fishing effort is understood to greatly varying degrees. Those areas with potentially the highest bycatch are those for which sampling is at low proportions of total effort, and consequently very high uncertainty exists (table 17). This is especially important for small surface longline vessels.
(b) The treatment of the very rare occasions when a large number of birds are caught in single fishing events remains a problem for robust statistical analysis as distributions classically used to model the data do not accommodate these occurrences well.
(c) The hierarchical nature of the sampling, i.e., observers are placed on vessels at the beginning of a trip, and observe all or most fishing events on that trip. Hence using tows/sets as the basic sampling unit may not be completely appropriate. The most serious consequence of not accounting for these aspects of the data in the model is that credible intervals will tend to be too narrow.

In summary, we consider that the methods and data requirements for the estimation of seabird bycatch in New Zealand fisheries remain an ongoing area for development. Our study has highlighted where the greatest uncertainties and deficiencies in the data are. Our confidence in the results

**TABLE 17**
**Observer coverage levels, fishing effort, and total predicted bycatch across fisheries in New Zealand in the 2004 fishing year**

| Fishing method[1] | Vessel size[2] | % fishing events observed | % total fishing events for 2004 | % total predicted bycatch |
|---|---|---|---|---|
| TR | Small | 0 | 44 | 0 |
| SLL | Small | 2 | 3 | 31 |
| BLL | Small | 1 | 10 | 5 |
| TR | Large | 10 | 39 | 46 |
| SLL | Large | 82 | <1 | 4 |
| BLL | Large | 12 | 4 | 13 |

[1] Bottom longline (BLL), surface longline (SLL) and trawl (TR) methods.
[2] Small = < 28 m; Large = >28 m.

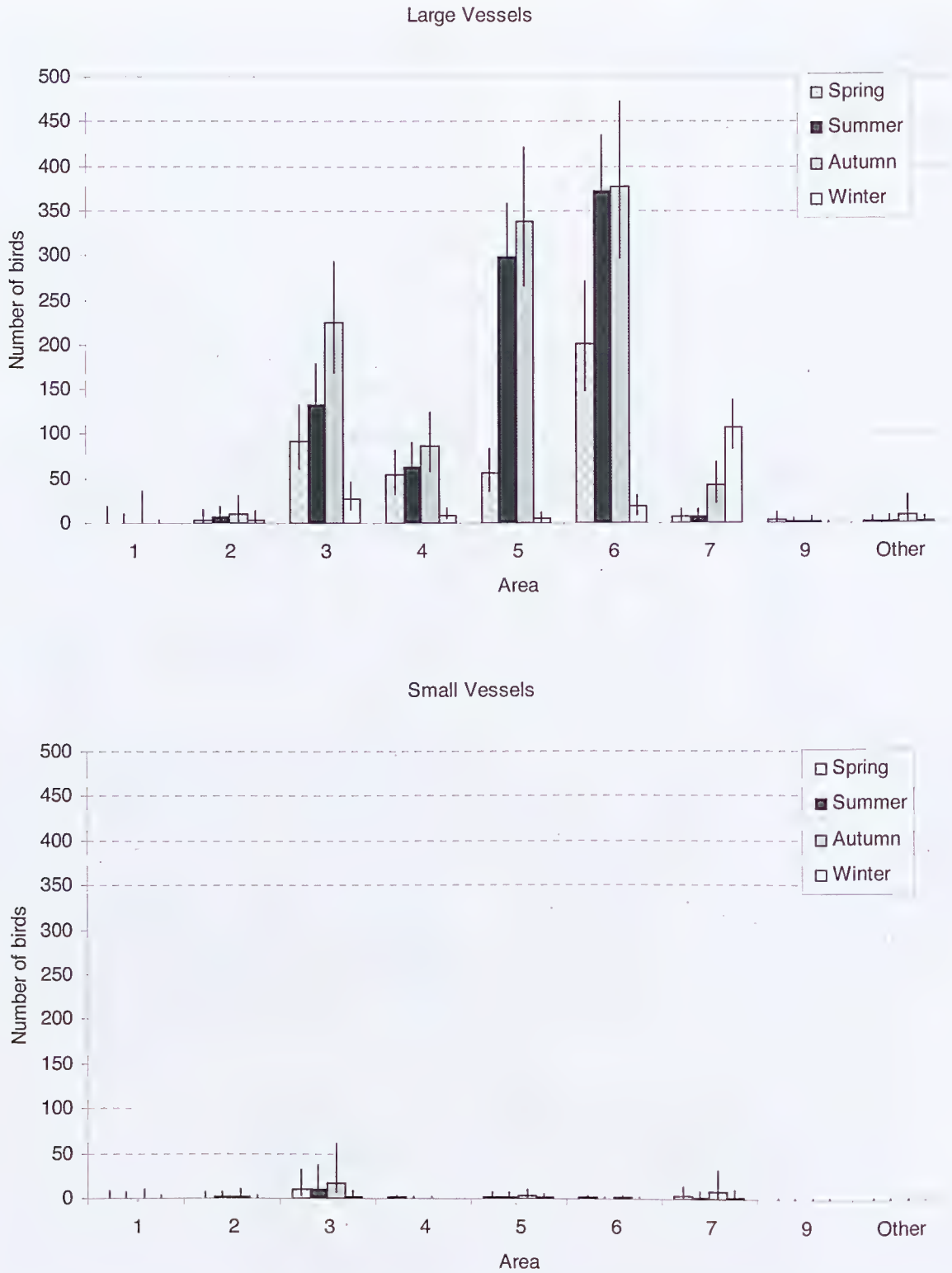


*FIG. 6 — The estimated seabird bycatch (number of birds) in New Zealand trawl fisheries during the 2004 fishing year, together with 95% credible intervals for large and small vessels (split at 28 m length), between seasons for different Fishery Management Areas.*

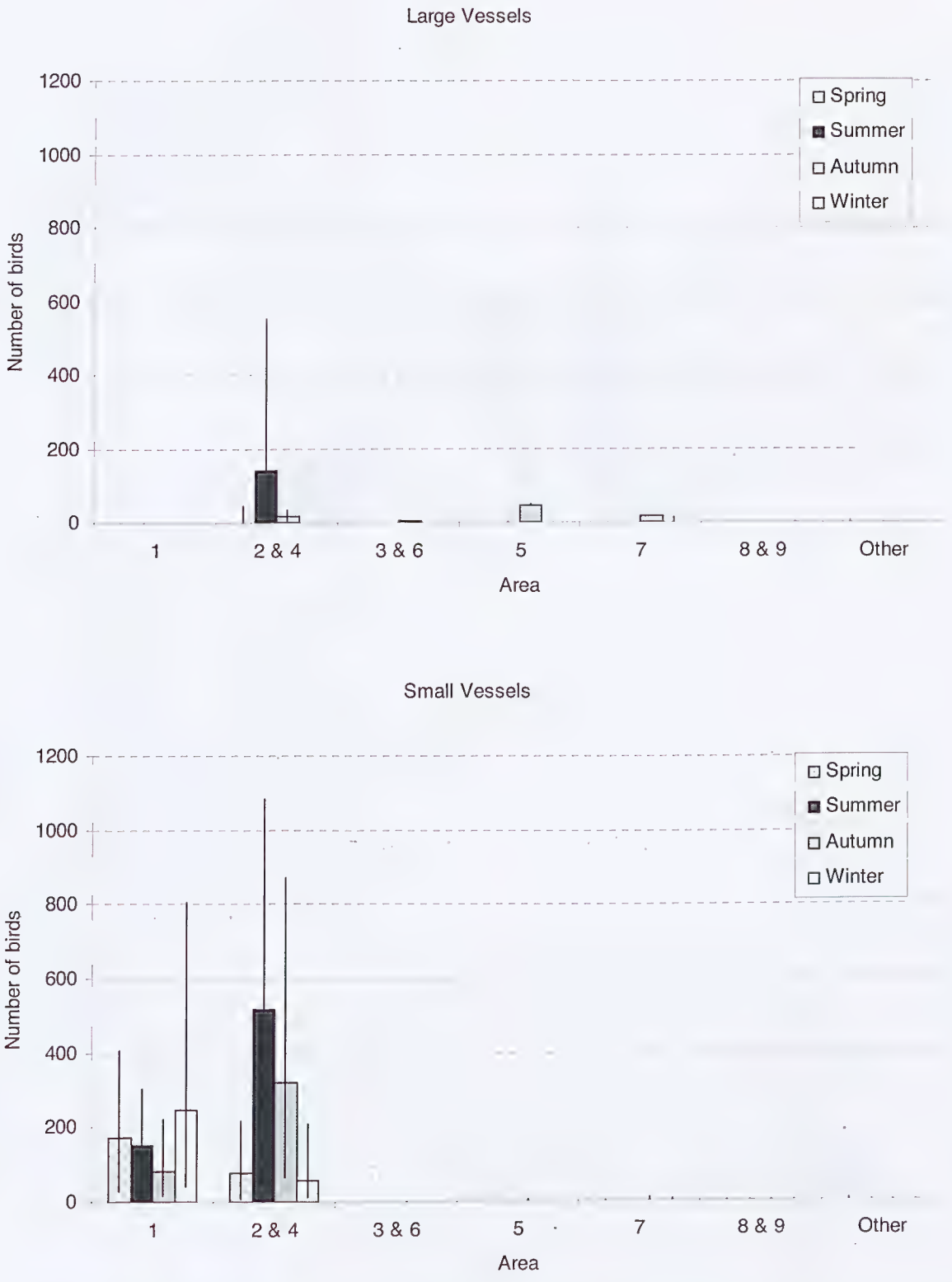


*FIG. 7— The estimated seabird bycatch (number of birds) in New Zealand surface longline fisheries during the 2004 fishing year, together with 95% credible intervals for large and small vessels (split at 28 m length), between seasons for different Fishery Management Areas.*

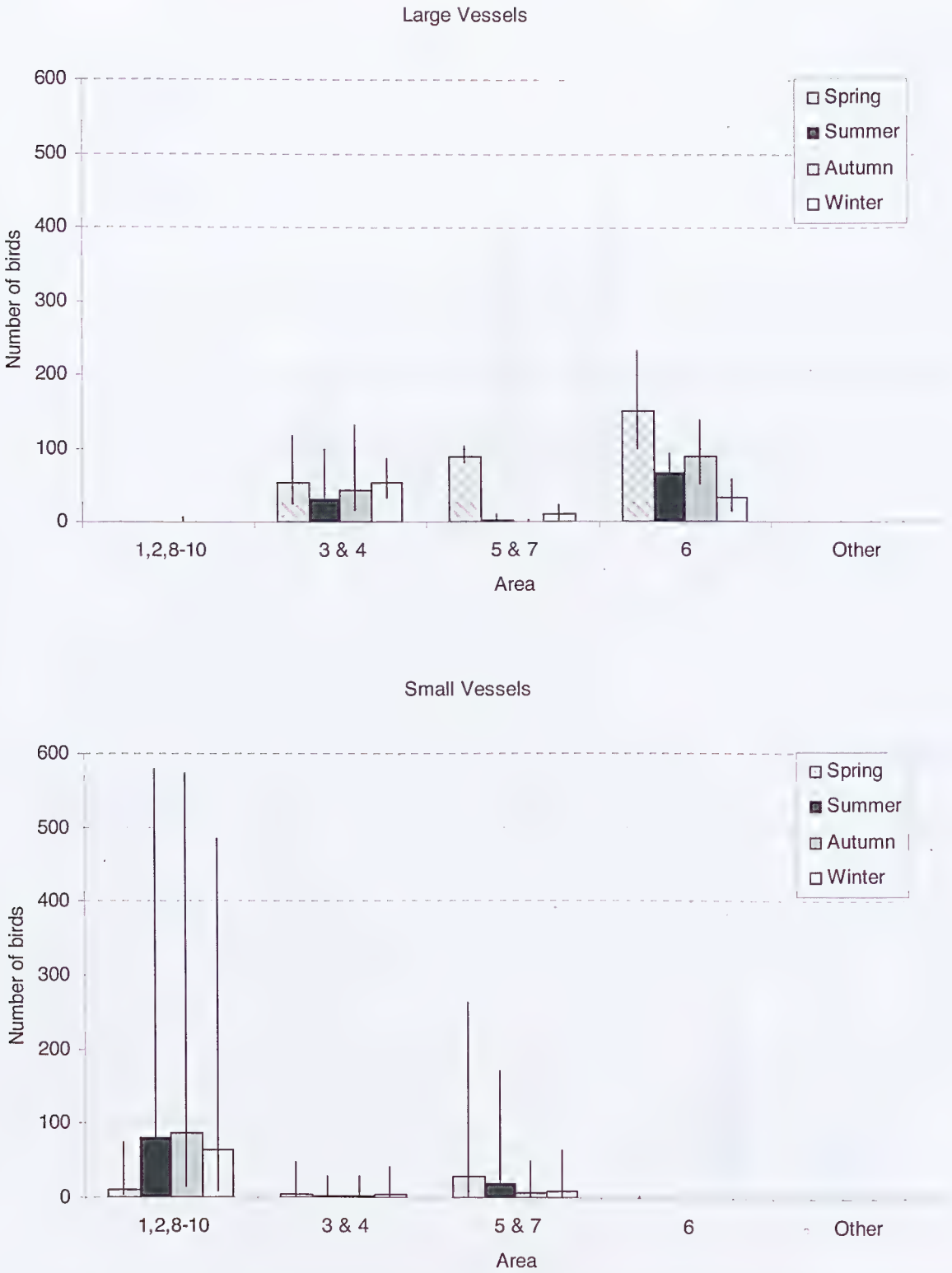


*FIG. 8 — The estimated seabird bycatch (number of birds) in New Zealand bottom longline fisheries during the 2004 fishing year, together with 95% credible intervals for large and small vessels (split at 28 m length), between seasons for different Fishery Management Areas.*

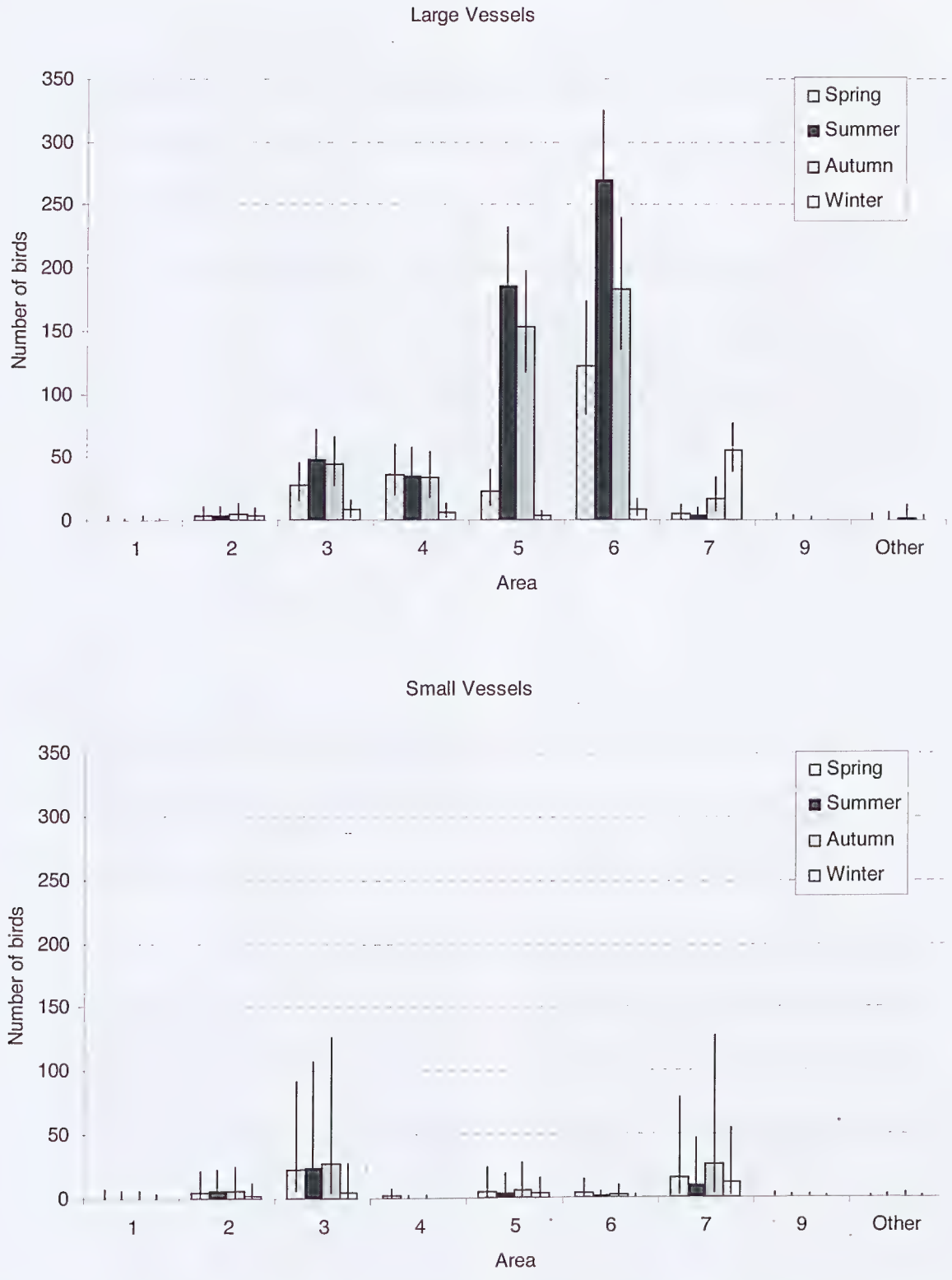


*FIG. 9 — The estimated albatross bycatch (number of birds) in New Zealand trawl fisheries during the 2004 fishing year, together with 95% credible intervals for large and small vessels (split at 28 m length), between seasons for different Fishery Management Areas.*

reported is somewhat limited by the matters discussed in this section. In particular, the wide confidence intervals for the estimates of seabird captures on small vessels across methods indicate that a far greater level of observer sampling would be required to gain precise and accurate estimates of captures for these fisheries.

## Estimation of New Zealand fisheries bycatch

### Observer coverage

Observer coverage across New Zealand fisheries was, on average, around 5% of the total fishing effort for the TR, SLL and BLL fisheries during 1998–2004. Coverage was concentrated in high-value fisheries or those with a high risk for stock sustainability (New Zealand Ministry of Fisheries 2007a), and on large vessels. These are mainly fishing for tuna, ling, and deep-water or middle-depths fish species such as Southern Blue Whiting, *Micromesistius australis* Norman, 1937, Hoki, squid, Orange-roughy, *Hoplostethus atlanticus* Collett, 1889, and oreos. Observer coverage in New Zealand fisheries is not deployed in a manner that facilitates examination of the environmental effects of fishing. In order to estimate more robustly seabird bycatch, we consider that there is a need for deployment of observers to cover a considerably higher proportion of all fishing effort, with observer coverage spread randomly across fleets, areas and seasons in relation to the likelihood of seabird capture (table 13). For the period of the study, observer deployment was not related to the risk of seabird capture or where fishing effort is deployed (table 17). Fishing effort by small vessels was observed at rates of less than 3%, even though they potentially catch 35% of the total seabirds caught and their effort comprised 57% of the total effort in 2004.

In order to understand species level effects of fishing, greater levels of observer coverage are required across all fishing effort, as the capture of individuals of any particular species is a rare event, requiring high sampling ratios to detect.

### Estimated total captures

For large vessels in New Zealand fisheries, seabird mortalities are likely to be approximately 3800 individuals annually, with approximately 2000 additional individuals captured on small vessels. There is high uncertainty associated with these estimates, particularly for mortalities associated with small vessels.

Previous estimates of bycatch in New Zealand fisheries have been focused on a small number of target fisheries, where observer coverage was greatest. This has allowed a detailed analysis of capture rates in specific fisheries, but not a comprehensive understanding of the likely scale of the bycatch problem to be established, and in particular where information is uncertain or lacking. Estimation of bycatch using a modelling approach has enabled prediction of where bycatch problems are most likely to be occurring, and has identified areas where a far greater level of observation and management of bycatch problems are necessary.

For fisheries from large TR vessels in 2004, it was estimated that approximately 2500 seabirds were caught in FMA3–7. Albatross bycatch for these areas for 2004 was estimated to be approximately 1300 birds. Previous estimates of seabirds captured in TR fisheries in New Zealand waters were limited to those identified during fishing for Hoki and squid fisheries, and for some areas only of these fisheries (Baird 2004a, b, c, 2005, Baird & Smith 2007). Bycatch totals previously reported were lower than indicated for large vessels in this study, with between 500 and 1700 seabirds estimated caught annually in TR fisheries where sufficient observer coverage was deployed between 1998 and 2004 (see the source materials for error estimation; Baird 2004a, b, c, 2005, Baird & Smith 2007). The differences in the results between the current study and previous work are not easy to reconcile. Although the primary data sets were the same, the grooming of data and inclusion of fishing effort in the different estimation procedures is likely to have differed considerably. This statement applies across all fishing methods discussed in this study.

Relatively few seabirds were caught on large SLL vessels, and the number has likely decreased through time. For 2004 it was estimated that approximately 230 were caught in FMA2–7, compared with approximately 650 for the same areas in 1998. Previous estimates of bycatch in these fisheries indicated that around 40–140 seabirds were caught on large SLL vessels for the fleet fishing under charter in New Zealand waters annually from 1998–2004, although these did not include all large SLL vessels fishing in New Zealand waters each year (Baird 2004a, b, c, 2005, Baird & Smith 2007).

For BLL fisheries, total seabird bycatch of approximately 630 for FMA3–7 was estimated for large vessels in 2004. The numbers of seabirds estimated caught has decreased through time, from a peak of approximately 2800 in 2001. Prior to 2001, problems with observer recording of data are likely to have resulted in inaccurate estimation, as the number of observed hooks was not reported, and unobserved birds returned by crew were counted in the bycatch totals. Estimated bycatch from BLL vessels from previous studies (Baird 2004a, b, c, 2005, Baird & Smith 2007) indicated levels of around 1400 reducing to around 208 seabirds per year from 2001 to 2004.

## Management of New Zealand fisheries seabird bycatch

In 2004, New Zealand implemented its National Plan of Action – Seabirds (NPOA-Seabirds) (New Zealand Department of Conservation 2004) which largely relied on voluntary codes of practice to implement effective mitigation strategies. Streamer lines were required as a mandatory measure for surface longline vessels. In 2005, following captures of large numbers of albatross in the squid trawl fishery, the Minister of Fisheries called for a review of the NPOA-Seabirds, and indicated that it should include a greater range of mandatory measures, and bring New Zealand management of seabird bycatch closer to international best practice. In August 2008, New Zealand continued to revise its NPOA-Seabirds, but has yet to announce a revised framework. Between 2005 and 2008, a range of mandatory mitigation measures was introduced, including trawl-warp scaring devices and the requirement to hold offal for half-hour periods for trawl vessels over 28 m, the use of streamer lines, line weighting or night setting in all longline fisheries (New Zealand Ministry of Fisheries 2006, 2007c, 2008b, c). In July 2008, the Minister of Fisheries announced that the requirement for offal management on trawl vessels was to be revoked due to vessel safety concerns (New Zealand Ministry of Fisheries 2008a).

The bycatch levels in the New Zealand trawl and longline fisheries reported in this study are similar to those observed in Antarctic BLL fisheries for toothfish *Dissostichus* sp., prior to the onset of rigorous management measures under

the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR) (Croxall *et al.* 2007, Waugh *et al.* 2008). In the CCAMLR BLL fishery in areas where the CCAMLR Conservation measures are required to be implemented (which excludes the French sub-Antarctic EEZ areas) seabird bycatch decreased from around 6500 birds in 1996 to zero birds in 2007. This example demonstrates that management of fisheries mortality of seabirds is feasible and achievable within realistic time frames.

The introduction of a range of mitigation requirements for New Zealand fisheries, and particularly those in 2008, represents a significant step towards reducing seabird bycatch in New Zealand waters. It is important to note, however, that the additional resources required for ensuring compliance with the measures and to increase observer coverage have not been allocated. These factors may weaken the New Zealand Government's abilities to either monitor bycatch levels or ensure compliance with the new mandatory requirements. The ongoing lack of a formal framework for seabird bycatch management, such as through a NPOA-Seabirds, also threatens to hamper New Zealand's ability to keep abreast of the continual improvement in bycatch mitigation strategies, such as through offal managment and line weighting currently under research through many trawl and longline fisheries worldwide.

### Trends in bycatch estimates

Seabird bycatch totals have remained low and stable through time for SLL fishing, for large vessels. It has decreased from around 2800 bids per year to around 600 birds per year for large BLL vessels. For large TR vessels however, there has been an increase in seabird mortality during the 1998–2004 fishing years, to around 2500 birds in 2002–2004, of which around one half was albatross. For all methods for small vessels, we do not have sufficient confidence in the estimates produced to attribute trends.

## CONCLUSIONS

Our study indicates that there is an ongoing level of seabird bycatch in New Zealand fisheries, which merits further monitoring, research and management. Approximately 5500 seabirds may be killed in New Zealand fisheries each year. For large vessels, approximately 3800 seabirds are caught. There were increasing numbers of seabirds estimated for TR fisheries between 1998 and 2004, to a total of approximately 2500 seabirds, of which around one half were albatross in 2004. Large numbers of albatross, and species of threatened status which occur across the fisheries are captured in proximity to their breeding areas, increasing the risk of population effects from the fisheries removals. Highest priority areas for monitoring and management action include fisheries by large TR vessels in southern areas (FMA3–7), small SLL vessels in northern and central areas (FMA1, 2 and 4), and for all BLL fishing.

To better understand the effects of fishing on the species observed caught in the New Zealand fisheries, further monitoring, management and research is warranted. This could include increased observer effort, a re-deployment of observer effort in areas with greatest risk of seabird capture, and detailed demographic studies of the species most at risk due to vulnerable population status. Estimation of fisheries bycatch of seabirds remains an important tool for the management of the effects of fishing on environmental sustainability. Tracking the evolution of bycatch levels across areas, target fisheries and years is pivotal to allowing targeted mitigation efforts, in areas where it is likely to reduce seabird bycatch. Due to the high conservation value of these species, their iconic status in society and the high level of endemism of the species known to be affected by TR, SLL and BLL fisheries, the management of seabird bycatch in New Zealand should be a high priority.

## ACKNOWLEDGEMENTS

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# PLASTIC INGESTION BY FLESH-FOOTED SHEARWATERS, *PUFFINUS CARNEIPES*, AND WEDGE-TAILED SHEARWATERS, *PUFFINUS PACIFICUS*

by Ian Hutton, Nicholas Carlile and David Priddel

(with two text-figures)

**Hutton, I., Carlile, N. & Priddel, D.** 2008 (31:x): Plastic ingestion by Flesh-footed Shearwaters, *Puffinus carneipes*, and Wedge-tailed Shearwaters, *Puffinus pacificus*. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 67–72. ISSN 0080-4703. Lord Howe Island Tours, PO Box 157, Lord Howe Island, NSW 2898, Australia (IH*), Department of Environment and Climate Change (NSW), PO Box 1967, Hurstville, NSW 2220, Australia (NC, DP). *Author for correspondence. Email: ianhutton@clearmail.com.au

The population of Flesh-footed Shearwaters, *Puffinus carneipes*, on Lord Howe Island, Australia, has decreased over recent decades. Known threats include long-line fishing and loss of nesting habitat. The recent occurrence of plastic debris in breeding colonies has raised concerns that plastic ingestion also may be contributing to the decline of this species. In this paper we investigate the extent of plastic ingestion by Flesh-footed and Wedge-tailed shearwaters, *Puffinus pacificus*, on Lord Howe Island. The remains of failed Flesh-footed Shearwater fledglings contained substantial quantities of plastics: up to 37 $cm^3$, equivalent to at least 31% of proventricular capacity. Road-killed adults (n = 21) had no plastic in their proventriculus. Proventricular contents of near-fledged birds, obtained by non-lethal means, showed that 79% of Flesh-footed Shearwaters and 43% of Wedge-tailed Shearwaters contained plastics, in volumes of up to 18.0 $cm^3$ and 2.8 $cm^3$, respectively. Plastic loads were significantly less in Wedge-tailed Shearwaters, the difference possibly due to different densities of plastic within the foraging locations of each species. The impact of plastic ingestion on the survival of Flesh-footed Shearwater chicks and fledglings, and the consequent impacts on the demography of the population are unknown and warrant further investigation.

**Key Words: marine pollution, marine litter, Lord Howe Island, Wedge-tailed Shearwater, plastic ingestion, Flesh-footed Shearwater, *Puffinus carneipes*, *Puffinus pacificus*.**

## INTRODUCTION

The Flesh-footed Shearwater, *Puffinus carneipes* Gould, 1844, is a pelagic seabird that breeds in dense colonies on islands in New Zealand and in southwestern Australia, as well as on St Paul and Lord Howe islands (Fullagar *et al.* 1974, Roux 1985, Ross *et al.* 1996, Taylor 2000). The species is a trans-equatorial migrant that winters on both sides of the North Pacific Basin (Marchant & Higgins 1990). The population of Flesh-footed Shearwaters on Lord Howe Island, estimated to be approximately 17 500 pairs, has decreased substantially during the past few decades (Priddel *et al.* 2006). Known causes of decline include mortality in long-line fishing (Baker & Wise 2005) and loss of nesting habitat (Priddel *et al.* 2006). Recent observations of skeletal remains containing sizable amounts of plastic, together with plastic debris scattered throughout the breeding colonies of the Flesh-footed Shearwater, raised concerns that plastic ingestion may also be contributing to the decline of this species (Hutton 2004). In this paper we investigate the prevalence of plastic ingestion by Flesh-footed and Wedge-tailed shearwaters, *Puffinus pacificus* (J.F. Gmelin, 1789), on Lord Howe Island. These two species breed concurrently during the Austral summer with eggs of both species typically laid in late November or early December and young fledging in late April and May (Fullagar *et al.* 1974, Priddel *et al.* 2006). Both species feed on, or close to, the surface, although Flesh-footed Shearwaters can also dive deeply. The Wedge-tailed Shearwater is common throughout the tropical and sub-tropical waters of both the Pacific and Indian oceans. The current size of the population on Lord Howe Island is not known; the most recent estimate, in 1972, suggested there were approximately 30 000 pairs (Fullagar *et al.* 1974).

Many species of seabirds are particularly susceptible to plastics in the marine environment. Small pieces of plastic floating on or near the surface of the ocean can resemble the adults and larvae of crustaceans (Mauchline 1980, Raymont 1983) or may harbour the eggs of pelagic fish (Colton *et al.* 1974). Consequently, seabirds mistake plastic for food and ingest it, sometimes in large quantities (Moser & Lee 1992, Laist 1997). Plastic ingestion has a range of lethal and sub-lethal consequences. By decreasing the capacity of the proventriculus, the accumulated plastic can lower food intake, resulting in lower fledging mass and decreased survival (Connors & Smith 1982, Ryan 1987a, Sievert & Sileo 1993). Plastics can impair gastrointestinal function or cause physical damage to the bird's gastrointestinal tract, including perforation, blockage or ulceration (Day 1980, Pettit *et al.* 1981, Zonfrillo 1985, Fry *et al.* 1987, Pierce *et al.* 2004). Also, during digestion plastic can release toxins such as polychlorinated biphenyls (Ryan *et al.* 1988), which can accumulate in the tissues of the bird causing impaired fitness and reduced survival (Pettit *et al.* 1981, van Franeker 1985). These, and other toxins, can also disrupt reproduction by lowering steroid hormone levels and delaying ovulation (Peakall 1970, Hoffman *et al.* 1996).

Adults of many species of seabirds habitually regurgitate indigestible parts of their food, such as squid beaks, fish bones and plastic (e.g., Hays & Cormons 1974). Chicks, however, tend not to regurgitate until they are almost fully fledged, so plastics accumulate in their gut during the nestling period. For both Flesh-footed and Wedge-tailed shearwaters this period is about 14 weeks (Roberts *et al.* 1975, Powell *et al.* 2007). The low propensity of chicks to regurgitate indigestible material places them at greater risk than adults. The physical presence of plastic in the gut not only prevents chicks from receiving a full load of food at each feed, but by

preventing proventricular contraction — an important cue in the stimulation of hunger — they may have depressed appetite and diminished feeding activity (Sturkie 1965, Connors & Smith 1982, Ryan 1988a). The lower nutrient intake may result in reduced fitness or death, often through dehydration (Sievert & Sileo 1993).

Despite these consequences, definitive evidence of death or physiological damage resulting from plastic ingestion is difficult to obtain (Auman *et al.* 1998), with comparative studies often failing to reach statistical significance because of small sample sizes (Day 1980, Connors & Smith 1982). Additionally, proving cause and effect is problematic because of the difficulty in isolating this particular factor (Ryan 1987a). In this study, we aim only to determine the extent of plastic ingestion in two species of shearwater, not its impacts.

## STUDY SITE

Lord Howe Island (31°30'S, 159°05'E) is a small (1455 ha) oceanic island in the South Pacific Ocean, 580 km east of the Australian mainland. An area of lowlands, much of which has been cleared for agriculture and settlement, separates the northern hills (209 m) from the two southern mountains (875 m). The main island is surrounded by a number of smaller islets and rocks. Flesh-footed Shearwaters nest on the main island at several locations within the lowlands, on calcarenite soils, in burrows up to 3.4 m long (Priddel *et al.* 2006). Wedge-tailed Shearwaters are more abundant, and their nesting grounds are more dispersed. They breed at numerous sites on the coastal fringe of the main island and on most of the smaller islets within the group.

## METHODS

Periodically, between October 2001 and August 2002, the colony of Flesh-footed Shearwaters near Neds Beach (see Priddel *et al.* 2006) was searched for the remains of dead shearwaters. Carcasses were collected and dissected. Any plastics were separated from the skeletal material and other remains. The number of pieces and approximate size of each piece were recorded. The plastic recovered from each bird was then submerged under water in a graduated flask and its volume determined (to the nearest 0.1 $cm^3$) by displacement. Small, trace amounts were recorded as 0.1 $cm^3$.

During 27–29 April 2005, Flesh-footed Shearwater colonies were searched at night to capture and sample live, near-fledged birds on the surface. These birds were approximately 13–14 weeks old and, judging from the development of their primaries, were about 1–10 days from fledging. We sampled the stomach contents of 56 birds by flushing the proventriculus with water, as described by Wilson (1984). This non-lethal technique generally recovers 89–100% of the proventricular contents (Ryan & Jackson 1986), so some plastic fragments may remain in the gut (Sileo *et al.* 1990). In an attempt to recover the entire stomach contents, each bird was flushed three times. The maximum volume of water pumped into the bird before it regurgitated was recorded and used to approximate the maximum capacity of the proventriculus. All regurgitated material was collected. The quantities of water and oil were measured using a graduated flask and then discarded. Any plastic or other solid material was retained and sorted. The volume of this material was later measured by water displacement. Moribund or malformed birds that were incapable of flight were flushed and then euthanased. These birds, together with any carcasses found, were dissected and the contents of the proventriculus removed, sorted and measured.

Between late April and early May 2005, Flesh-footed Shearwater colonies were searched by day to collect freshly dead birds (n = 6) and regurgitated boluses (n = 124) containing plastic and other indigestible material. Boluses were collected only if fresh (i.e., the mucus in them was still wet). The volumes of plastics and other solid material were measured as described above.

Flesh-footed Shearwaters found killed on the roads between 2001 and 2007 were collected and stored frozen. These were subsequently thawed and dissected. The contents of the proventriculus were removed and the volume of plastic was measured.

On 29 April 2005, 30 near-fledged Wedge-tailed Shearwater chicks were extracted from their burrows and their proventricular contents sampled using the methods described above. Judging from the development of their primaries, these birds were approximately 2–3 weeks from fledging. After all fledglings had departed the colony, the nesting area was searched to locate the remains of dead chicks and regurgitated boluses of plastic and other indigestible material.

### Statistical analyses

Single-factor ANOVAs were used to examine differences in (1) the mean quantities of plastic in live and dead Flesh-footed Shearwater chicks, (2) the quantity of plastic in chicks and in boluses, and (3) the amount of plastic and other indigestible material. Differences in the quantity of plastic flushed from the two species of shearwater were examined by single-factor ANOVA, after data were log-transformed to normalise variances. All means are given ± SD.

## RESULTS

### Flesh-footed Shearwaters

The skeletal remains of 14 Flesh-footed Shearwaters were found in the area searched during 2001–02. All these remains contained plastic, in volumes ranging from 10–37 $cm^3$ (mean 20.5 ± 7.1 $cm^3$). In total, 579 pieces of plastic were recovered from these 14 birds, an average of 41 pieces per individual. The pieces were predominantly flat fragments of containers and other articles that ranged in size from about 25 $mm^2$ to 1500 $mm^2$; with a median size of about 30 x 20 mm. Identifiable plastic items in carcasses included bottle, carton or tube caps (16), bag or balloon ties (5), strapping tape (2), a golf tee (1) and a pen top (1).

Of the 56 near-fledged Flesh-footed Shearwater chicks sampled during 2005, 44 (79%) contained plastic (mean 2.6 ± 3.6 $cm^3$; range 0.1–18.0 $cm^3$, fig. 1). Forty-one birds (73%) contained oil, in volumes ranging from 0.5–38.5 ml (mean 9.9 ± 9.1 ml). Thirty-one birds (55%) contained other solids (squid beaks, otoliths, pumice and fish bones) in volumes ranging from 0.1–5.0 $cm^3$ (mean 0.7 ± 1.0 $cm^3$). Maximum proventriculus volume was 120 ml. For birds that contained some indigestible material, the mean quantity of plastic in the proventriculus exceeded the total amount of other solid

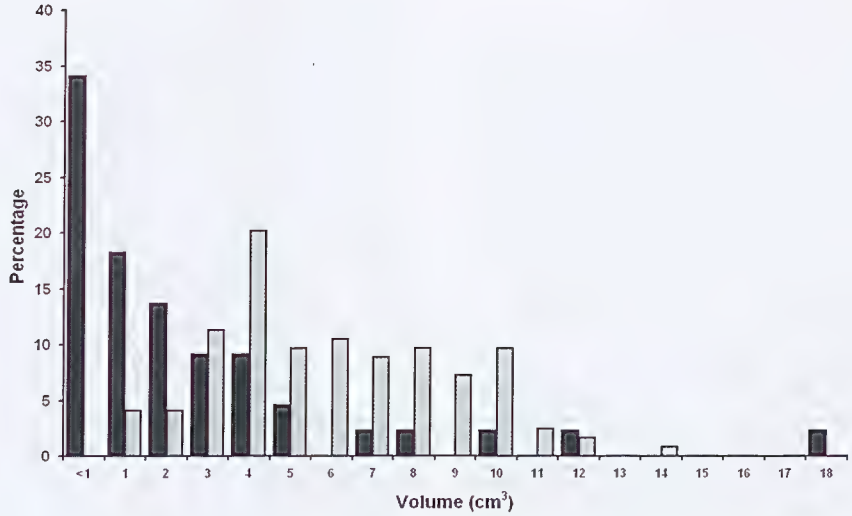


*FIG. 1 — Frequency distribution of the volume of plastic contained in the proventriculi of near-fledged Flesh-footed Shearwater chicks (solid bars) and in regurgitated boluses of indigestible material (light bars). Data include only those birds (n = 44) and boluses (n = 124) that contained plastic.*

material (ANOVA, $F1_{,73} = 8.635$, $P < 0.004$).

One of the 56 birds sampled had malformed wings and, although it was otherwise healthy, it was unable to fly, and so was euthanased. The proventricular contents of this bird obtained by flushing consisted of one squid beak (0.1 cm³) and one plastic bead (0.1 cm³). Necropsy found the proventriculus to be empty indicating that all proventricular contents had been removed by flushing. Another two individuals were moribund when found; these birds were flushed but were omitted from the sample of 56 above because they were clearly unwell and may not have been feeding normally. Both were euthanased. Flushing yielded 20 small pieces of plastic (1.6 cm³) and a small amount of other solids (0.1 cm³) from one bird, and nothing from the other. Necropsy found one additional piece of plastic (0.1 cm³) in the proventriculus of the first bird and nothing in the second, indicating that flushing removed 94% of the proventricular contents.

Six other near-fledged chicks from the 2005 cohort were found dead. Necropsy revealed that each bird contained significant quantities of plastic (mean 24.5 ± 5.0 cm³, range 18–32 cm³). Dead birds contained significantly more plastic than did live birds (ANOVA, $F1_{,48} = 178.9$, $P < 0.0001$).

All 124 freshly regurgitated boluses that were collected contained plastic (mean 5.8 ± 2.8 cm³; range 1.0–14.0 cm³, fig.1) and small amounts of other indigestible solids. All boluses also contained various amounts of down, indicating they had been regurgitated by chicks, and consistent with observations that most adults had departed the colony by this time. The mean quantity of plastic in each bolus exceeded the mean amount of plastic flushed from each bird (ANOVA, $F1_{,166} = 36.887$, $P < 0.0001$; birds without plastic were excluded).

A total of 21 adult Flesh-footed Shearwaters were found dead on the road after having been killed by vehicles. No plastic was found in any of these birds.

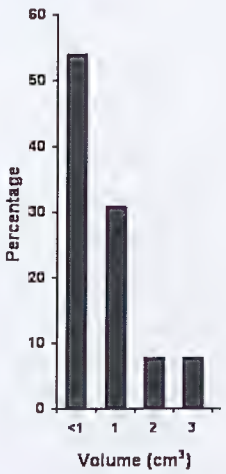


*FIG. 2 — Frequency distribution of the volume of plastic contained in the proventriculi of near-fledged Wedge-tailed Shearwater chicks. Data include only those birds (n = 13) that contained plastic.*

## Wedge-tailed Shearwaters

Of 30 Wedge-tailed Shearwater chicks sampled for proventricular contents during 2005, 13 (43%) contained plastics (mean 0.7 ± 0.9 cm³; range 0.1–2.8 cm³, fig. 2). Ten birds (33%) contained other indigestible material (fish bones, squid beaks, pumice and wood) in volumes ranging from 0.1–4.5 cm³ (mean 2.0 ± 1.2 cm³). All birds contained varying amounts of fish and squid, but none contained oil. Maximum proventricular volume was 90 ml. The quantity of plastic found in Wedge-tailed Shearwater chicks was less than that recovered from Flesh-footed Shearwater chicks (ANOVA, $F1_{,53} = 4.541$, $P = 0.038$, figs 1 and 2).

Of 22 Wedge-tailed Shearwater carcasses found in the colony, three (14%) contained plastic (1.5 cm³, 5.0 cm³ and 11.0 cm³). No boluses of regurgitated plastic were found during searches of Wedge-tailed Shearwater colonies.

## DISCUSSION

This study has shown that both Flesh-footed and Wedge-tailed shearwaters breeding on Lord Howe Island ingest plastic. Plastic ingestion was particularly prevalent among Flesh-footed Shearwaters, and many unsuccessful fledglings had died with substantial amounts of plastic in their proventriculus. The highest plastic load (37 $cm^3$) was equivalent to 31% of proventricular capacity of the largest individual. Based on these observations we speculate that some Flesh-footed Shearwater chicks probably died as a result of excessive quantities of ingested plastic. However, we know of only one instance where death of a chick can be directly attributed to ingested plastic. Necropsy of this chick, found dead in 2005, revealed that the proventriculus had been punctured by a sharp piece of plastic (S. Thalmann pers. comm.).

Seventy-nine per cent of near-fledged Flesh-footed Shearwater chicks on Lord Howe Island contained plastic. The quantity of plastic recovered from live chicks was highly variable, but generally was less than that recovered from dead chicks: whereas 50% of dead birds contained >5 $cm^3$ of plastic, only 10% of live birds contained this amount. It is possible that flushing failed to remove all plastic from the proventriculus, thus accounting for the apparent difference in plastic loads between live and dead birds. However, evidence from the small number of birds that were both flushed and necropsied indicated that less than 6% of items remained in the proventriculus after flushing.

The appearance of numerous boluses containing regurgitated plastic throughout the Flesh-footed Shearwater colonies in May of each year, after most adults had left, indicated that many fledglings offloaded the plastic that had accumulated in their gut before they departed from the island. The amount of plastic recovered from live birds was less than that contained in these boluses and it is likely that many of the birds sampled had already regurgitated some plastic. We conclude that plastic ingested by Flesh-footed Shearwater chicks accumulates in the proventriculus until shortly before fledging, when the young emerge from their burrows and regurgitate their proventricular contents, presumably in preparation for flight. The absence of plastic in adults indicates that these birds expel any plastic they ingest, at least some of it being passed onto their offspring.

Although 43% of Wedge-tailed Shearwater chicks on Lord Howe Island contained plastic, plastic ingestion was much less of a problem for this species (see figs 1 and 2), at least during the period of this study. Compared to Flesh-footed Shearwater chicks of approximately the same age, (i) fewer individuals contained plastic, (ii) lesser quantities of plastic were present in the proventriculus of those birds affected, and (iii) boluses containing plastic were not obvious in the colony. The reasons for these species-related differences are unclear. Although the feeding behaviour of the two species differs, Wedge-tailed Shearwaters tend to forage closer to the surface than do Flesh-footed Shearwaters and thus would be more, rather than less, likely to encounter floating plastics. It is possible that Wedge-tailed Shearwaters are much more selective in the items that they pick up, but a more likely explanation is that they forage in areas where there are lower concentrations of plastics. Proventricular oil was found in 73% of the Flesh-footed Shearwater chicks sampled, but in none of the Wedge-tailed Shearwater chicks. This difference reflects dissimilarity in the foraging range of the two species, with Flesh-footed Shearwaters probably travelling further from Lord Howe Island than do Wedge-tailed Shearwaters (Schultz & Klomp 2000). During winter, Wedge-tailed Shearwaters forage at much lower latitudes in the North Pacific than do Flesh-footed Shearwaters (Marchant & Higgins 1990) and species that winter in the tropics tend to have much lower plastic loads than those that winter further north, presumably because the density of plastic in the tropical Pacific is lower than in the North Pacific (Spear *et al.* 1995). Although plastics are ubiquitous throughout all the oceans of the world (Derraik 2002), variability in ocean circulation patterns can change the amount of marine debris regionally and therefore the amount available to seabirds foraging in a particular area (e.g., Morishige *et al.* 2007).

The high incidence of plastic in Flesh-footed Shearwater chicks on Lord Howe Island was not unusual for this species or other congeners. Of 83 Flesh-footed Shearwaters killed in driftnet fisheries in the North Pacific during 1990–91, 79 (95%) contained ingested plastic (Robards *et al.* 1997). In the same study, the incidence of plastic ingestion was 85% (n = 543) for Sooty Shearwaters, *Puffinus griseus* (J.F. Gmelin, 1789), and 88% (n = 200) for Short-tailed Shearwaters, *P. tenuirostris* (Temminck, 1835). A similar study by Ogi (1990) found comparable levels of plastic ingestion: 89% (n = 193) for Sooty Shearwaters and 82% (n = 265) for Short-tailed Shearwaters. Reported incidences of plastic ingestion by Wedge-tailed Shearwaters, although highly variable, have been somewhat lower: 3–29% (Sileo *et al.* 1990); 12–60% (Spear *et al.* 1995); 60% (Fry *et al.* 1987); 43% (this study).

From a sample of Wedge-tailed Shearwaters shot at sea in the central Pacific between 1984 and 1991 Spear *et al.* (1995) found the incidence of plastic varied among age classes: 20% of fledglings, 60% of pre-breeding adults and 12% of post-breeding adults contained plastic. The relative incidence of plastic in each age class is consistent with our interpretation of data obtained from Flesh-footed Shearwaters on Lord Howe Island, i.e., plastic, much of which is picked up at the winter foraging grounds, is offloaded to chicks by adults and then offloaded by chicks via boluses prior to fledging. Ryan & Jackson (1987) and Ryan (1988b) also reported lighter plastic loads in adults after the breeding season compared to before it, and suggested that offloading of plastics by parents to young accounted for this difference. Spear *et al.* (1995) argued that the low incidence of plastic in fledglings at sea offered little evidence to support this hypothesis, but these authors failed to take into account that, as shown in this study, many fledglings purge themselves of plastic before departing from the breeding grounds.

On Lord Howe Island Flesh-footed Shearwaters breed among human habitation, so the absence of any previous local record of plastic ingestion by this species suggests that this phenomenon may have become more prevalent in recent years. Other studies, however, have shown that the level of plastic ingestion can vary dramatically between years (Ryan 1988b, Sileo *et al.* 1990). For example, plastic was present in 44% of Red-tailed Tropicbird, *Phaethon rubricauda* Boddaert, 1783, chicks at Midway Atoll, Northwestern Hawaiian Islands, in 1986 but in only 2% of chicks the following year (Sileo *et al.* 1990). Similarly, the mean volume of plastic ingested by Laysan Albatross, *Phoebastria immutabilis* (Rothschild, 1893), chicks in consecutive years was 46 $cm^3$ and 5 $cm^3$ (Sileo *et al.* 1990). Further studies are needed to assess the long-term trends in plastic ingestion by Flesh-footed Shearwaters on Lord Howe Island.

Although the incidence of plastic varies considerably both among species and among individuals within species (Furness 1985, Azzarello & Van Vleet 1987, Ryan 1987b, Moser & Lee 1992, Spear *et al.* 1995) the growing body of evidence suggests that the frequency and quantity of ingested plastic in seabirds is increasing (Robards *et al.* 1997). More than one-third of the world's seabird species are now known to ingest plastic, with Procellariiformes being among those most affected (63% of species, Laist 1997). In many species, the proportion of individuals that contain plastic exceeds 80%. This increase in plastic ingestion parallels an observed increase in plastic pollution in the areas where many seabirds forage (Moser & Lee 1992). The increase in plastic ingestion in seabirds is a disturbing trend, particularly as this is only one of a number of anthropogenic threats affecting these species (Baker *et al.* 2002). Long-line fishing and loss of nesting habitat have both been identified as serious threats to the population of Flesh-footed Shearwater on Lord Howe Island (Baker & Wise 2005, Priddel *et al.* 2006). The threatened status of this population warrants further studies be undertaken to monitor population size, along with experimental research or modelling to assess the impact of plastic ingestion on chick and fledgling survival.

Many countries have taken comprehensive action to address the issue of marine litter through legislation, enforcement of international agreements, provision of reception facilities for ship-generated wastes, adoption of improved practices for the disposal of waste on land, and support for extensive beach clean-up activities, as well as information, education and public awareness programs (UNEP 2005). Despite these efforts, the situation, globally as well as regionally, is worsening; the quantity of plastic waste circulating in the world's oceans continues to grow (Derraik 2002). It is clear that the problem cannot be solved solely by means of legislation, law enforcement and technical solutions, it is also a cultural problem that requires changes to attitudes, behaviours and management approaches (UNEP 2005).

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# AT-SEA DISTRIBUTION OF SATELLITE-TRACKED GREY-FACED PETRELS, *PTERODROMA MACROPTERA GOULDI,* CAPTURED ON THE RUAMAAHUA (ALDERMEN) ISLANDS, NEW ZEALAND

by Catriona J. MacLeod, Josh Adams and Phil Lyver

(with three text-figures and five tables)

**MacLeod, C.J., Adams, J. & Lyver, P.** 2008 (31:x): At-sea distribution of satellite-tracked Grey-faced Petrels, *Pterodroma macroptera gouldi*, captured on the Ruamaahua (Aldermen) Islands, New Zealand. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 73–88. Landcare Research, PO Box 40, Lincoln 7640, New Zealand (CJMacL*, PL); United States Geological Survey, Western Ecological Research Center, Moss Landing Marine Laboratories, 8272 Moss Landing Road, Moss Landing, California, 95039, USA (JA). * Author for correspondence. Email: macleodc@landcareresearch.co.nz

We used satellite telemetry to determine the at-sea distribution of 32 adult (non-breeders and failed breeders) Grey-faced Petrels, *Pterodroma macroptera gouldi*, during July–October in 2006 and 2007. Adults captured at breeding colonies on the Ruamaahua (Aldermen) Islands ranged across the southwestern Pacific Ocean and Tasman Sea between 20–49°S and 142°E and 130°W. Petrels were located almost exclusively over offshore waters >1000 m depth. The extent of their distributions was similar across years, but petrels ranged farther south and west in 2006. Individuals displayed a high degree of spatial overlap (48–62% among individuals) and area use revealed three general "hotspots" within their overall range: waters near the Ruamaahua Islands; the central Tasman Sea; and the area surrounding the Chatham Rise. In July–August 2006, most petrels congregated over the Tasman Sea, but for the same period in 2007 were predominantly associated with Chatham Rise. The home ranges of petrels tended to overlap disproportionately more than expected with the Australian Exclusive Economic Zone and less than expected with High Seas, relative to the area available in each zone, in July–August 2006. Accordingly, multiple nations are responsible for determining potential impacts resulting from fisheries bycatch and potential resource competition with Grey-faced Petrels.

**Key Words: satellite telemetry, utilisation distribution, home range, foraging range, Grey-faced Petrels, *Pterodroma macroptera gouldi.***

## INTRODUCTION

Seabirds of the order Procellariiformes have evolved life history traits, behaviours and physiologies well-suited for coping with food resources that are patchily distributed over enormous (millions of square kilometres) pelagic habitats. Their ability to fast for extended periods and their extremely low cost-of-flight enables individuals to traverse large oceanic expanses while searching for patchily distributed food (Warham 1990, 1996). Our current knowledge about the distribution of gadfly petrels (*Pterodroma* spp.) at sea is limited to results from observations of birds from ships (Sladden & Falla 1928, Hyrenbach *et al.* 2006) and indirect measures of chick diet composition (Imber 1973). The relative importance of different oceanographic features and environmental conditions (e.g., bathymetry, ocean productivity, and wind speed) for determining gadfly petrel distributions at sea is not well known (but see Haney 1986, 1987, Stahl & Bartle 1991, Spear *et al.* 2001).

Detailed information about *Pterodroma* spp. ranges, migration routes, and habitat associations is required to understand environmental factors influencing species abundance and conservation issues. Increasing miniaturisation of tracking devices now enables continuous monitoring of small-bodied (<1000 g) procellariiform seabirds moving in remote and inaccessible areas of the ocean (e.g., Shaffer *et al.* 2006, Adams & Takekawa 2008). Tracking seabird movements can provide insights into patterns in range utilisation associated with elevated biological productivity ("hotspots"), significant biogeographical features such as oceanic current fronts, sea mounts, and sea-ice extent (Spear *et al.* 2001, Clarke *et al.* 2003, Morato *et al.* 2008), and intensively fished areas (Phillips *et al.* 2006). This information can be valuable for managers making decisions about species conservation or modifications to certain natural resource uses and developments within significant habitats (e.g., implementation of fisheries quotas and bycatch restrictions; Lewison *et al.* 2004). Awareness of hotspots will help managers prioritise their environmental management to times and places that are most important for species' persistence (i.e., critical foraging areas).

Our study focuses on the at-sea distribution of the Grey-faced Petrel, *Pterodroma macroptera gouldi* (Hutton, 1869), a culturally significant species to the northern *iwi* (Māori tribes), e.g., Hauraki, Ngāti Awa, Ngāti Wai) of New Zealand (B. Hughes pers. comm. 2006, T. Renata pers. comm. 2007, Lyver *et al.* 2008, T. Shortland pers. comm. 2008). The Grey-faced Petrel mainly breeds on islands off the east coast of northern New Zealand, with the largest colonies on the Ruamaahua (Aldermen), Moutohora (Whale) and Whakaari (White) islands (Wodzicki & Robertson 1959, Imber 1976, Heather & Robertson 2000). Populations of conspecific Great-winged Petrel, *Pterodroma macroptera macroptera* (A. Smith, 1840), nest on islands adjacent to the south coast of Western Australia, in the South Atlantic (e.g., Tristan de Cunha Group, Gough, Kerguelen and Crozet islands) and in the South Indian Ocean (e.g., Kermedec Islands; Warham 1956, Richardson 1984, Cuthbert & Sommer 2003). *Pterodroma m. gouldi* (and *P. m. macroptera*) are unique among the gadfly petrels as they are winter-time (Austral) breeders that lay eggs during June and July. Imber (1973) suggested that this phenology may be an adaptation to maximise nocturnal feeding time and more effectively exploit diel vertically migrating prey (e.g., various cephalopods and myctophid fishes). Observations of *P. macroptera* subspecies at sea indicate they rarely congregate in groups, except when resting on the sea surface during

the day (Imber 1973, Hyrenbach *et al.* 2006). Historic accounts report that Grey-faced Petrels ranged mainly to the east of New Zealand between 31° and 42°S and about 150 km off the coast to 3200 km east (145°W) during the breeding season (Fleming 1950).

Here, we present the at-sea distribution patterns (i.e., movements) among individual Grey-faced Petrels captured during the breeding season at two breeding colonies on the Ruamaahua Islands during 2006 and 2007 and outfitted with satellite transmitters. The aims of our study were to determine: (a) the extent of the at-sea distribution of Grey-faced Petrels; (b) temporal variation in the degree of spatial overlap among individuals and the locations of high-use areas (hotspots); and (c) the extent and variability in the use of the New Zealand and Australian Exclusive Economic Zones (EEZs) versus the High Seas.

## METHODS

### Satellite telemetry

We attached satellite transmitters (Sirtrack Kiwisat 202 Platform Transmitter Terminals [PTTs], Havelock North, NZ) to 32 adults during two breeding seasons (2006 and 2007; table 1). Each year, we captured eight petrels at the colony during two periods: incubation (16–17 July 2006 and 19–20 July 2007) and early chick-rearing (12–13 September 2006 and 16–17 September 2007). In 2006, petrels were captured on the surface at two different locations (Locations 1, 2) on Ruamaahuanui in July and a single location (Location 3) on a neighbouring island, Hongiora, in September (fig.1, table 1). The reproductive status of the petrels in 2006 was, therefore, unknown because individuals captured were not associated with a burrow; petrels ultimately were classified as non-/failed breeders. In 2007 during the pre-laying period (April), we fitted 20 artificial nesting chambers (hereafter chambers) to existing burrows at a single location (Location 4) on Ruamaahuanui (fig. 1) to enable us to identify adult petrels of known breeding status. In July 2007, we captured eight incubating petrels from chambers and outfitted them with PTTs. In September 2007, only two chambers housed incubating petrels, so the remaining six adults were captured on the surface of the colony at night (similar to 2006) at Location 4 on Ruamaahuanui (fig. 1; table 1). Elsewhere, records indicate that most viable eggs hatch before 10 September (Bethells Beach breeding colonies; G. Taylor, pers. comm.). Thus, in 2007, we had eight confirmed breeders in July and two confirmed failed breeders and six petrels of unknown breeding status in September. We determined sex for 24 of the 32 individuals outfitted with PTTs using a DNA extraction technique adapted for feathers (Lambert *et al.* 2000).

We weighed petrels (±5 g) using a Pesola® spring scale (Pesola Ag, Baar, Switzerland) and fitted each with a uniquely numbered metal band. PTTs were affixed to

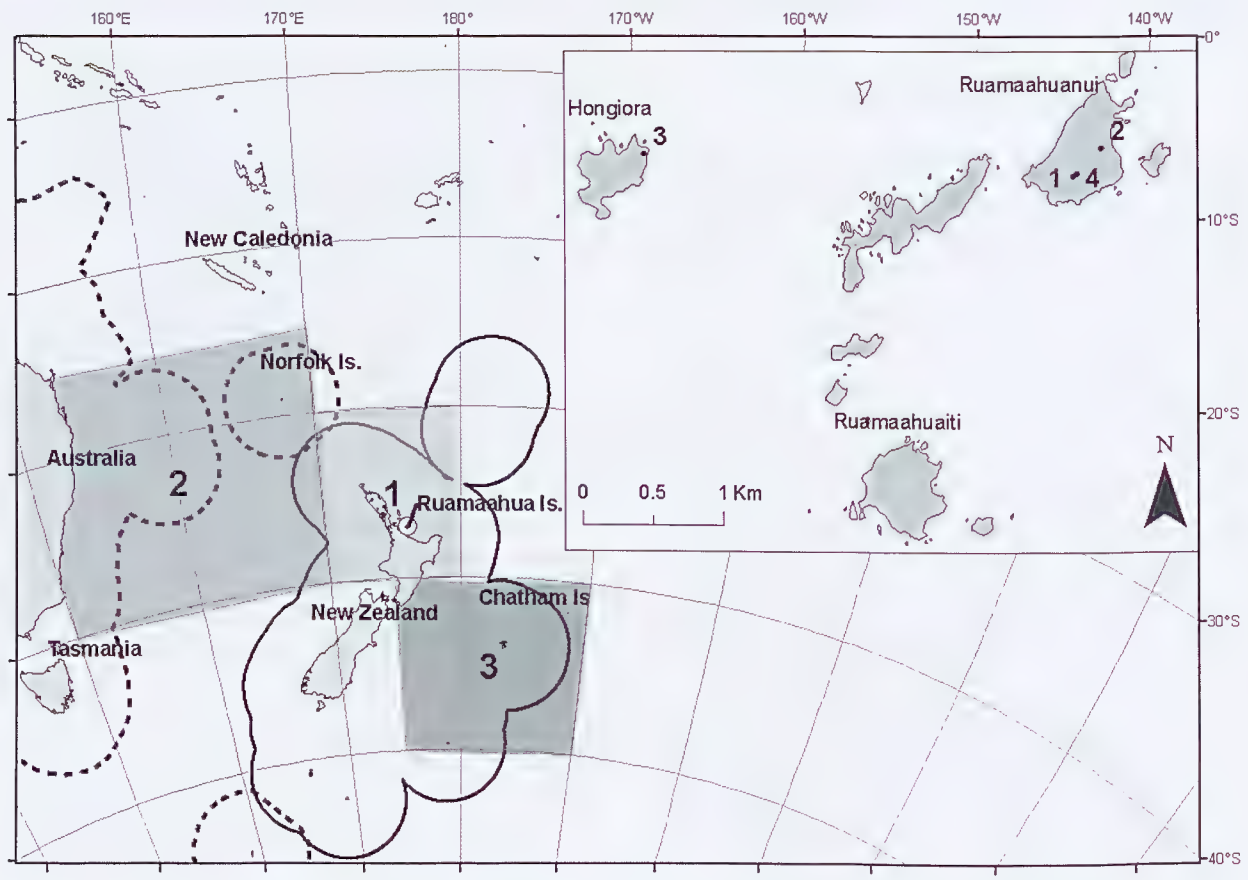


*FIG. 1 — Key locations in the oceanic region surrounding New Zealand, including the location of the breeding colonies on the Ruamaahua Islands. A more detailed map of the Ruamaahua Islands is presented in the inset which shows the four locations where the birds were captured and released for satellite tracking. Solid black line indicates the New Zealand Exclusive Economic Zone and the dashed black line indicates the Australian Exclusive Economic Zone in the main figure and the three shaded boxes indicate the maximal extent of hotspots identified in the UD analysis (1 = Ruamaahua, 2 = Tasman, 3 = Chathams hotspots).*

**TABLE 1**
**Summary of the capture and satellite tracking data for 32 Grey-faced Petrels tagged at breeding colonies on the Ruamaahua Islands, New Zealand**

| Bird identity | Body mass (g) | Sex | Capture information | | | | Days tracked | Locations retained | Distance travelled (km) | Latitudinal range | | Longitudinal range | | Home range area (km²) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Island[1] | Location | Position | Date | | | | North | South | East | West | 25% contour | 50% contour | 95% contour |
| 588 | 540 | U | RN | 2 | Surface | 16 Jul 2006 | 64 | 605 | 21,736 | 23.4°S | 39.2°S | 176.5°W | 151.4°E | 97,281 | 308,448 | 1,381,941 |
| 599 | 540 | U | RN | 1 | Surface | 16 Jul 2006 | 77 | 729 | 26,964 | 28.7°S | 45.6°S | 176.6°E | 153.3°E | 52,812 | 182,817 | 1,078,353 |
| 601 | 515 | U | RN | 1 | Surface | 16 Jul 2006 | 84 | 779 | 28,363 | 27.9°S | 44.8°S | 176.1°E | 150.8°E | 110,079 | 312,174 | 1,222,938 |
| 589 | 605 | U | RN | 2 | Surface | 17 Jul 2006 | 65 | 601 | 19,960 | 22.8°S | 39.5°S | 177.8°E | 151.9°E | 62,856 | 222,021 | 1,138,293 |
| 590 | 500 | U | RN | 2 | Surface | 17 Jul 2006 | 71 | 697 | 25,544 | 26.4°S | 45.2°S | 168.0°W | 148.8°E | 80,109 | 275,643 | 1,498,905 |
| 592 | 530 | U | RN | 2 | Surface | 17 Jul 2006 | 41 | 385 | 14,224 | 23.0°S | 41.4°S | 177.0°W | 155.0°E | 50,382 | 193,914 | 1,031,049 |
| 596 | 530 | U | RN | 2 | Surface | 17 Jul 2006 | 72 | 675 | 31,804 | 24.3°S | 40.7°S | 154.0°W | 150.7°E | 184,680 | 587,088 | 2,416,473 |
| 597 | 570 | U | RN | 2 | Surface | 17 Jul 2006 | 27 | 264 | 10,274 | 25.9°S | 38.1°S | 178.9°E | 157.4°E | 66,015 | 189,054 | 766,179 |
| 595 | 560 | F | HA | 3 | Surface | 12 Sep 2006 | 70 | 660 | 19,395 | 28.1°S | 39.3°S | 1797°E | 151.0°E | 39,690 | 117,936 | 656,181 |
| 598 | 520 | F | HA | 3 | Surface | 12 Sep 2006 | 39 | 407 | 13,669 | 29.8°S | 43.2°S | 158.1°W | 156.9°E | 70,632 | 212,058 | 955,962 |
| 600 | 630 | F | HA | 3 | Surface | 12 Sep 2006 | 40 | 413 | 19,934 | 26.2°S | 48.7°S | 166.4°W | 160.3°E | 116,559 | 368,388 | 1,571,967 |
| 602 | 575 | M | HA | 3 | Surface | 12 Sep 2006 | 17 | 170 | 7,117 | 26.7°S | 38.4°S | 177.4°E | 151.7°E | 36,450 | 125,388 | 533,061 |
| 591 | 530 | F | HA | 3 | Surface | 13 Sep 2006 | 40 | 393 | 20,475 | 19.6°S | 41.8°S | 130.6°W | 162.7°E | 98,820 | 314,604 | 1,508,949 |
| 593 | 550 | M | HA | 3 | Surface | 13 Sep 2006 | 26 | 285 | 12,704 | 24.9°S | 40.4°S | 168.6°W | 153.5°E | 89,748 | 263,331 | 1,036,233 |
| 594 | 550 | M | HA | 3 | Surface | 13 Sep 2006 | 22 | 180 | 7,420 | 31.2°S | 43.1°S | 166.4°W | 176.0°E | 45,927 | 135,027 | 554,202 |
| 603 | 590 | M | HA | 3 | Surface | 13 Sep 2006 | 82 | 794 | 26,211 | 30.9°S | 47.8°S | 176.9°E | 144.1°E | 42,525 | 115,830 | 601,830 |
| 605 | 570 | M | RN | 4 | Burrow | 19 Jul 2007 | 41 | 409 | 14,719 | 24.0°S | 45.4°S | 166.7°W | 155.5°E | 54,270 | 193,104 | 1,074,465 |
| 606 | 580 | M | RN | 4 | Burrow[2] | 19 Jul 2007 | 89 | 871 | 31,935 | 22.4°S | 41.8°S | 149.6°W | 157.1°E | 101,007 | 323,676 | 1,570,347 |
| 611 | 700 | M | RN | 4 | Burrow | 19 Jul 2007 | 49 | 480 | 19,119 | 26.2°S | 46.4°S | 158.2°W | 158.1°E | 78,975 | 269,811 | 1,347,111 |
| 614 | 585 | M | RN | 4 | Burrow | 19 Jul 2007 | 79 | 900 | 28,506 | 25.5°S | 45.1°S | 153.7°W | 154.0°E | 77,274 | 256,041 | 1,525,068 |
| 604 | 600 | M | RN | 4 | Burrow | 20 Jul 2007 | 88 | 989 | 35,869 | 31.4° S | 48.9°S | 168.6°W | 160.9°E | 121,257 | 379,728 | 1,734,615 |
| 607 | 545 | M | RN | 4 | Burrow | 20 Jul 2007 | 14 | 73 | 3,354 | 32.1°S | 42.6°S | 174.0°W | 174.0°E | 29,565 | 90,477 | 328,131 |
| 612 | 550 | M | RN | 4 | Burrow | 20 Jul 2007 | 96 | 997 | 38,163 | 30.7°S | 47.9°S | 139.3°W | 141.5°E | 117,045 | 366,363 | 1,885,923 |
| 613 | 550 | M | RN | 4 | Burrow | 20 Jul 2007 | 73 | 678 | 27,076 | 23.6°S | 41.5°S | 157.6°W | 152.8°E | 96,795 | 379,890 | 1,879,605 |
| 608 | 530 | F | RN | 4 | Burrow | 16 Sep 2007 | 20 | 237 | 10,110 | 25.1°S | 41.4°S | 152.0°W | 176.1°E | 78,489 | 244,458 | 925,344 |
| 609 | 535 | M | RN | 4 | Surface | 16 Sep 2007 | 34 | 381 | 13,050 | 31.5°S | 37.2°S | 172.0°W | 156.6°E | 49,086 | 146,610 | 657,234 |
| 610 | 500 | F | RN | 4 | Surface | 16 Sep 2007 | 20 | 238 | 7,496 | 28.4°S | 37.0°S | 177.1°E | 160.5°E | 26,406 | 89,181 | 431,487 |
| 615 | 525 | F | RN | 4 | Burrow[2] | 16 Sep 2007 | 8 | 91 | 2,088 | 29.3°S | 37.0°S | 178.4°E | 167.6°E | 11,988 | 37,827 | 196,668 |
| 616 | 540 | F | RN | 4 | Surface | 16 Sep 2007 | 36 | 424 | 11,330 | 30.7°S | 38.6°S | 177.5°E | 158.8°E | 28,836 | 79,866 | 442,422 |
| 617 | 535 | F | RN | 4 | Surface | 17 Sep 2007 | 17 | 188 | 5,827 | 28.1°S | 36.9°S | 176.8°E | 160.5°E | 27,540 | 81,243 | 407,997 |
| 618 | 545 | F | RN | 4 | Surface | 17 Sep 2007 | 46 | 542 | 18,586 | 22.6°S | 40.8°S | 130.3°W | 174.5°E | 78,813 | 247,374 | 1,232,982 |
| 619 | 525 | F | RN | 4 | Surface | 17 Sep 2007 | 76 | 861 | 18,986 | 27.3°S | 37.0°S | 177.0°E | 150.7°E | 45,036 | 128,952 | 586,278 |

[1] RN = Ruamaahuanui, HA = Hongiora, see fig. 1 for capture locations. [2] Indicates birds that were tagged from the same burrow.

petrels between the scapulae over the central synsacral area to be close to the petrel's centre of mass (Healy *et al.* 2004) using a suture-glue-tape method modified after Newman *et al.* (1999). Specifically, each PTT was attached using four sterilised, monofilament, polypropylene surgical sutures (CP Medical, PolyproTM 2-0), a single strip of tape (Tesa® AG 4657, Hamburg, Germany) underlying approximately four feathers, and a few drops of glue (TDS Loctite® 422, Henkel Technologies, Düsseldorf, Germany) between the base of the PTT and approximately four surface feathers. We opted for this attachment to maximise tag retention throughout the battery life of the transmitter (approximately 60 days). More importantly, the method provided a more stable attachment than tape alone and reduced the risk of the tag lifting and straining the feathers (and thereby altering the petrel's centre of mass) during normal flight, when the petrel rolled from side to side or pursued prey on the wing. An identical attachment has been successfully used with Sooty Shearwaters, *Puffinus griseus* (J.F. Gmelin, 1789), Greater Shearwaters, *Puffinus gravis* (O'Reilly, 1818), and Hawaiian Petrels, *Pterodroma sandwichensis* (Ridgway, 1884) (J. Adams unpubl. data). PTTs weighed 29–34 g (~5% of the mean mass among all petrels outfitted with PTTs; mean ± SE; 554.7 ± 7.1 g) and were 62 mm long × 26 mm wide × 17 mm high. PTTs were programmed to transmit signals with a 60-s repetition rate with 2-h "on:off" duty cycles to maximise transmission duration and the number of "high-quality" locations obtained during "on" periods.

Geographical locations of individual petrels were determined by ARGOS (CLS America 2007) and archived using the online Satellite Tracking and Analysis Tool (STAT; Coyne & Godley 2005). Prior to analysis we used the following procedure to filter potentially spurious locations provided by ARGOS. We retained all location classes (LC 3, 2, 1, 0, A, B) before filtering. For each satellite pass, ARGOS calculates two candidate PTT locations (i.e., "mirror locations" result from the intersection of two cones of satellite reception and the platform altitude ellipsoid) and specifies which location is the accurate one (CLS America 2007). Occasionally ARGOS will select the incorrect "mirror location" (i.e., <1% of total locations per individual; CLS America 2007). So we used STAT to identify potentially incorrect "mirror locations" in the raw data and manually swapped any incorrect mirror locations. Second, we applied a 0.001-h time filter with STAT to identify duplicate location records (i.e., two locations recorded at the same time). The less accurate (determined by the location classes provided by ARGOS) of the two duplicate records were then removed. Pre-filtered data were exported from STAT and read into Program R (version 2.6.1; R Development Core Team 2007). In R, we used the speed-distance-angle ARGOS filter (*SDAfilter* in the argosfilter package version 0.5; Freitas *et al.* 2008) with maximum ground speed threshold set to 60 km $h^{-1}$ and default distance and angle threshold values. Filtered data then were read into MATLAB (MathWorks 2006) where we used purpose-built functions to create hourly, linear interpolated locations along each individual petrel's track-line (Tremblay *et al.* 2006). We did not interpolate track-line segments between endpoints separated by ≥8 h to avoid gross misrepresentation of an individual's actual position. Interpolation provided a temporally uniform distribution of locations for analyses that, unlike the raw ARGOS locations, are not biased by satellite orbital parameters and the petrel's latitudinal position (Georges *et al.* 1997, BirdLife International 2004).

## Estimating utilisation distributions

To define the spatial probability distribution for each petrel, we calculated the fixed-kernel utilisation distribution (UD, Van Winkle 1975) for each individual's interpolated trackline using a raster composed of 9 × 9 km grid cells (81 $km^2$). This grid cell size was selected because 9 km is a conservative distance that encompasses the average accuracy of the combined ARGOS location classes. Nicholls *et al.* (2007) report a mean accuracy of <1 km for LC 3, 2, and 1 and <5 km for LC 0, A, and B. To calculate kernel values, we fitted a bivariate-normal model (using the kernelUD function, adehabitat v.1.7.1 package in R; Calenge 2006) and specified a conservative neighbourhood smoothing parameter of 27 km to best represent the actual tracks of individual petrels given the accuracy of the ARGOS locations (see Hyrenbach *et al.* 2002). All other parameters were set using the kernelUD default values. In this analysis, the home range of each individual was defined as the area within the 95% probability density contour (White & Garrott 1990; hereafter 95UD area). For clarity, "UD" alone refers to the fixed-kernel utilisation distribution (i.e., continuous probability surface composed of 9 × 9 km grid cells).

## Estimating spatial overlap among individuals

To quantify spatial overlap among individuals within a given time period (i.e., two bimonthly periods [July–August and September–October] in each year), we calculated the percentage UD overlap of each petrel with all other individuals tracked during the same time period. Here we used the "PHR" method (Fieberg & Kochanny 2005) in the *kernel.overlap* function (Calenge 2006) to compute the probability of finding petrel $j$ in the home range of animal $i_1$, $i_2$ etc., where the home range of each animal $i_n$ was defined by its UD within its 95UD area. For each individual, we estimated the mean percentage UD overlap with all other individuals tracked during the same period. We considered this metric to be an informative, quantitative measure of the degree to which individuals within a population (i.e., the set of tracked petrels) share space in conjunction with the graphical analyses used to quantify hotspots (described below).

## Comparative analysis of tracking and distribution parameters

We compared bimonthly (July–August and September–October) estimates of the number of days petrels were tracked, latitudinal and longitudinal range estimates and UD overlap among individuals in each year. For each of these variables, we fitted a linear mixed effects model (*lme* in R using the *nlme* package v. 3.1-86; Pinheiro & Bates 2000) with the following explanatory variables: "bimonthly period", "year", and the "bimonthly period × year" interaction. Where appropriate, we controlled for variation in sampling effort, by including the number of days that petrels were tracked in the model as a covariate. Because the same individuals were tracked in more than one bimonthly period, we included the petrels' unique PTT identities as a random effect in the model to control for repeated measures. We identified the minimum adequate model (i.e., determined whether or not to retain explanatory variables in the model) using a backward stepwise process. All percentage estimates of UD overlap were arcsin-transformed prior to analysis (Zar 1996). We also compared three parameter estimates (the number of days tracked, distance travelled and 95UD area) for males and females using unequal variance

t-tests. All analyses were carried out using the statistical package, R (R Development Core Team 2007).

### Quantifying hotspots of distribution at sea

For the purposes of our study, we define hotspots as those areas where the probability of locating an individual petrel (i.e., its UD) was high and where several individuals also visited the same location. To identify hotspots within a given bimonthly period in each year, we first summed the UD surfaces of the relevant individuals' 95UD areas ($\Sigma_n$ 95UD). We then weighted the summed UD values for each 9 × 9 km grid cell according to the number of unique petrels that occupied it, using the formula: $\Sigma_n$ 95UD/$n_t^{-1}$, where $n_t$ was the number of unique individual 95UD areas which overlapped the grid cell. Thus, $\Sigma_n$ 95UD/$n_t^{-1}$ estimates for grid cells occupied by only a few petrels were down-weighted relative to those occupied by a greater number of petrels. To distinguish between areas with different probabilities of detecting occupancy by Grey-faced Petrels, we divided the weighted $\Sigma_n$ 95UD estimates into quartiles; the uppermost quartile indicated a very high probability of detecting occurrence and, therefore, the hotspots. For each bimonthly-year period, we also quantified the proportion of individuals that occupied the three main hotspots (fig. 1) and the proportion of the total area of individual 95UD areas that overlapped with each hotspot.

### Quantifying use of Exclusive Economic Zones and High Seas

The EEZs for Australia and New Zealand include the oceanic waters extending to 200 nautical miles from each nation's mean low water mark. High Seas refer to those ocean areas outside a specific nation's EEZ. These waters and their resources are open to use by any nation, and do not belong to any nation. Rights and responsibilities for international use of the High Seas are defined under the United Nations Convention on the Law of the Sea (UNCLOS) to which both New Zealand and Australia are signatories (United Nations 2007).

We estimated the proportions of the individual 95UD areas within Australia and New Zealand's EEZ and High Seas. We then used compositional analysis (CA, Aebischer *et al.* 1993) to determine whether or not the 95UD areas were randomly located with respect to these zones. To do this, we identified the area of each zone potentially available (i.e., "available area") for each petrel. We assumed that the available area for each petrel was all oceanic habitat within an area defined by a radius equivalent to the maximum straight-line distance between each petrel's initial location record and all other location records obtained for that petrel within the relevant bimonthly-year time period. The initial location may have been either on the Ruamaahua Islands or at sea, depending on the time period considered. We then calculated the proportion of each petrel's available area that intersected with each zone. To determine whether petrels were disproportionately located within one zone versus another, we compared the relative proportion of the available area within each zone to the relative proportions of the 95UD areas (i.e., "used areas") within each zone. Our CA used the randomisation test in the *compana* function (Calenge 2006). All other parameters (the number of replicates, zero replacement values, and alpha thresholds) were set using the default values.

## RESULTS

### Satellite tracking

Individual Grey-faced Petrels were tracked for (mean ± SE) 51 ± 5 days (range: 8–96) and PTTs generated 10 ± 0.2 usable (i.e., post-filtered) locations per day (table 1). On average, individuals travelled 18 500 ± 1697 km (range: 2088–38 160) and had home ranges (95UD areas) which covered 1 068 000 ± 94 833 km$^2$. Core use areas (as indicated by the 25% and 50% UD area contours) covered 70 840 ± 6476 and 226 200 ± 20 942 km$^2$ respectively (table 1).

All petrels tagged early in the breeding season (July) were males (feather samples were only available for 2007). The sex composition of petrels tagged in September varied between years (in 2006: four males, four females; 2007: one male, seven females; table 1). Males were tracked for a slightly longer duration (males vs females; 55 ± 9 vs 38 ± 6 days), tended to travel farther (20 403 ± 3217 vs 13 445 ± 1959 km) and have larger home ranges (1 132 910 ± 154 860 vs 810 567 ± 140 535 km$^2$), but none of these trends was statistically significant (Unequal variance t-tests; Days: t = -1.61, d.f. = 21.3; $P$ = 0.122; Distance: t = -1.85, d.f. = 19.4; P = 0.08; Area: t = -1.54, d.f. = 22.0, P = 0.140).

### Colony attendance and breeding status

Five of 32 (16%) Grey-faced Petrels returned to the colony at least once (table 2). The flight path of one additional petrel (bird identity: 594) indicated that it likely also returned to the colony (absence of location data during the following 24-hour period upon near arrival prevented confirmation, but this pattern is consistent with the inability of satellites to receive signals when petrels are underground (J. Adams unpubl. data)). The number of individuals that returned to the colony varied between observation periods. Two of the eight petrels tagged in each July returned to the colony, but during September, only one returned in 2006 and none in 2007 (table 2). Patterns of colony attendance also varied among the five petrels that returned to the colony. All, except one (bird identity: 596), returned to the colony only once. Petrel 596 completed four trips over a five-day period with each trip lasting 1–3 days and covering 69–1244 km (table 2). The average duration of the trips undertaken by the remaining four petrels was 33 ± 6 days, during which they travelled an average total distance of 14 430 ± 2743 km and maximum straight-line distance from the colony of 996 ± 343 km (range: 41–2587). For petrels that returned to the colony, the mean number of outward- and inward-bound flights observed per individual was 3 ± 1 and 2 ± 1 respectively. Of the 11 artificial nesting chambers that had eggs during July 2007, none had chicks in September, indicating that all breeding attempts initiated in these chambers were unsuccessful, irrespective of whether the breeding petrels were tagged or not.

### At-sea distribution

Grey-faced Petrels tracked during the breeding season (July–December) covered a wide geographical range (table 1; latitudinal range: 19.6–48.9°S; longitudinal range: 141.5°E–130.3°W). In both years, a few individuals travelled north towards New Caledonia or east to the Kermadec Trench or beyond. During September–October each year, one or two petrels travelled as far south as Tasmania, travelling along the east coast of the island to its southern-most

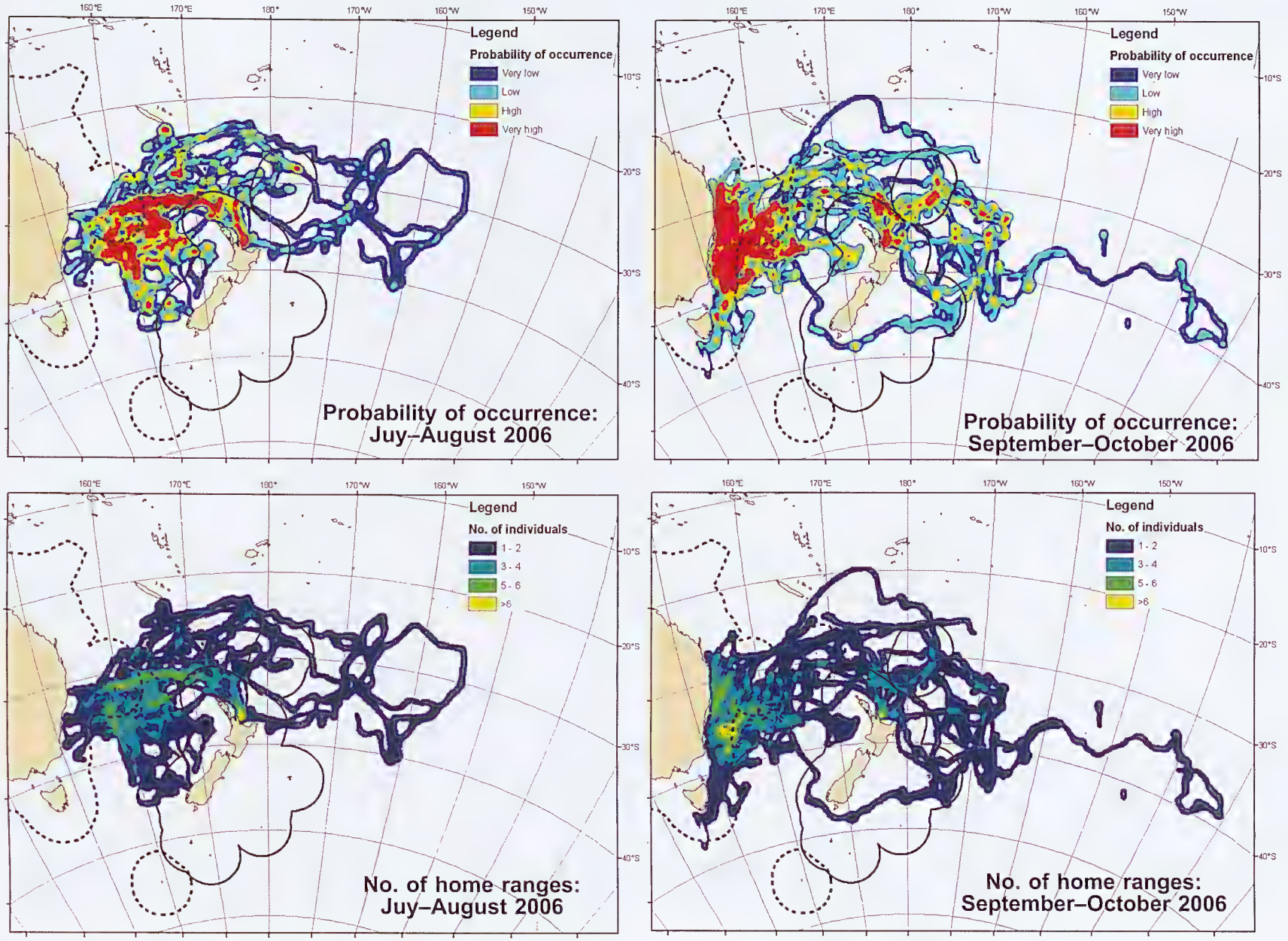


*FIG. 2 — At-sea distribution of satellite-tracked Grey-faced Petrels from the Ruamaahua Islands, New Zealand. Each shows the probability of occurrence and the number of home ranges which overlapped each 9 x 9 km grid cell within a given bimonthly period and year. Solid black line indicates the New Zealand Exclusive Economic Zone and the dashed black line indicates the Australian Exclusive Economic Zone.*

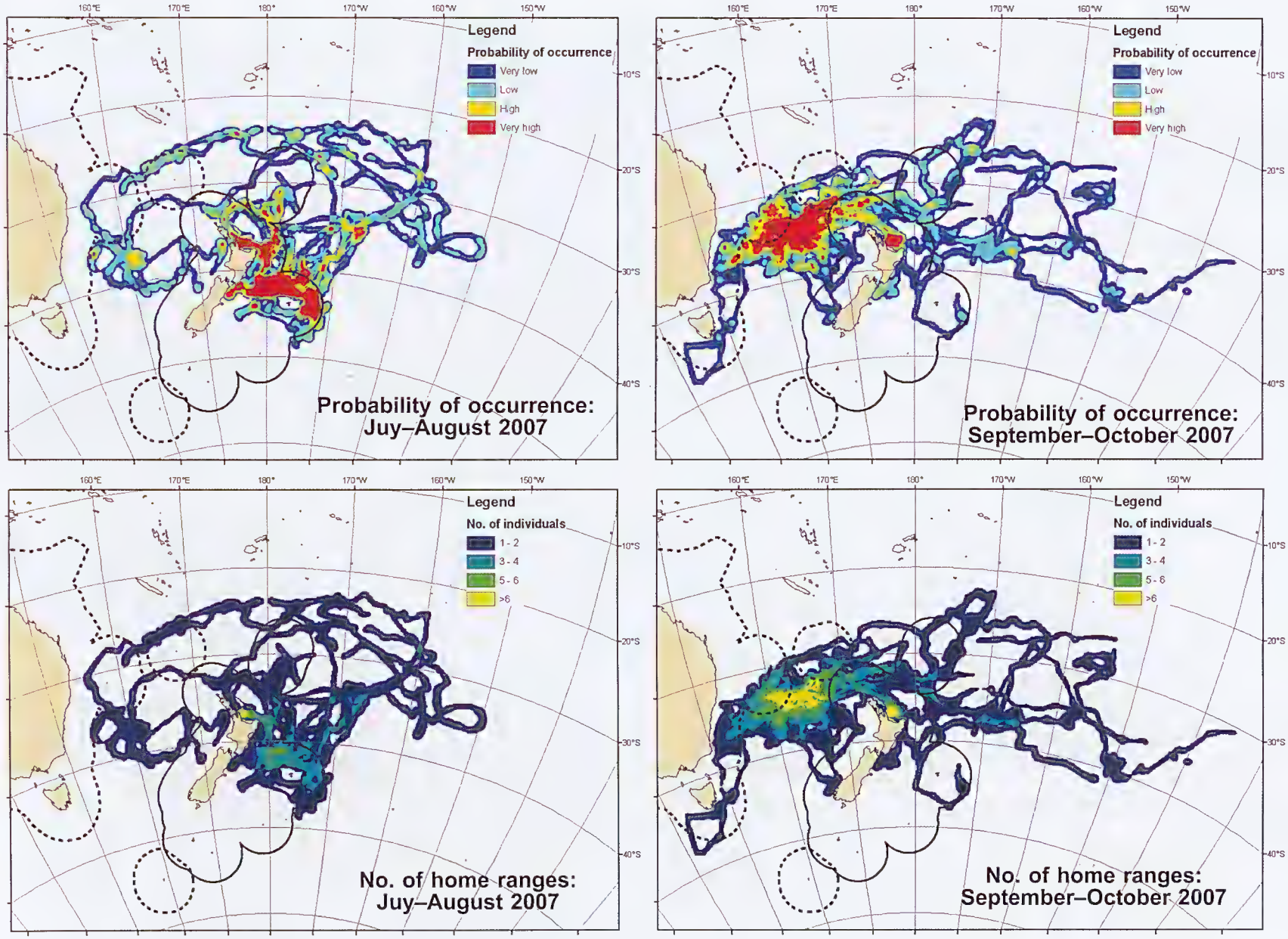


*FIG. 2 continued — At-sea distribution of satellite-tracked Grey-faced Petrels from the Ruamaahua Islands, New Zealand. Each shows the probability of occurrence and the number of home ranges which overlapped each 9 x 9 km grid cell within a given bimonthly period and year. Solid black line indicates the New Zealand Exclusive Economic Zone and the dashed black line indicates the Australian Exclusive Economic Zone.*

**TABLE 2**
**Summary of foraging trip information for satellite-tracked Grey-faced Petrels that returned to the colony at least once**

| Bird identity | Trip identity | Date start | Trip duration (days) | Return to colony | Distance travelled (km) | Maximum distance from colony (km) |
|---|---|---|---|---|---|---|
| 588 | 1 | 16 Jul2006 | 23 | Y | 9,339 | 1,681 |
| | 2 | 8 Aug 2006 | 41 | N | 13,347 | 2,257 |
| 596 | 1 | 17 Jul 2006 | 3 | Y | 1,244 | 494 |
| | 2 | 20 Jul 2006 | 1 | Y | 69 | 41 |
| | 3 | 21 Jul 2006 | 1 | Y | 86 | 41 |
| | 4 | 22 Jul 2006 | 1 | Y | 83 | 42 |
| | 5 | 23 Jul 2006 | 66 | N | 31,189 | 2,991 |
| 600 | 1 | 12 Sep 2006 | 23 | Y | 11,003 | 1,556 |
| | 2 | 5 Oct2006 | 17 | N | 9,796 | 1,681 |
| 604 | 1 | 24 Jul 2007 | 48 | Y | 21,547 | 1,524 |
| | 2 | 10 Sep 2007 | 36 | N | 16,358 | 1,415 |
| 611 | 1 | 31 Jul 2007 | 37 | Y | 15,830 | 2,587 |
| | 2 | 6 Sep 2007 | 12 | N | 4,111 | 1,589 |

All Grey-faced Petrels were originally captured on the Ruamaahua Islands' breeding colonies (see table 1 for details).

point, before heading north again. However, most petrels congregated within the Tasman Sea between 30 and 40°S (i.e., within the High Seas between Australia and New Zealand; fig. 2).

Although the extent of the Grey-faced Petrel geographical range was similar between years, there was notable temporal variation. Individual tracking durations varied between bimonthly and yearly periods (table 3). After controlling for the variation in tracking duration (i.e., number of days) between periods, the southern latitudinal extent was consistent across years for the July–August period, but varied between years in the September–October period, with petrels travelling farther south in 2006 than in 2007 (table 3). Grey-faced Petrels also ranged farther west in 2006 than in 2007, when individuals travelled parallel to the edge of the southeast Australian continental shelf, and later in the season in both years. There was no evidence of a significant shift in the northern extent of their distribution, but there was a trend for petrels to travel farther east in 2007, when a greater proportion congregated around the Chatham Rise (fig. 2).

## Spatial overlap and hotspots of distribution at sea

Grey-faced Petrels occurred primarily over waters >1000 m deep (mean ± SE; 93 ± 2% of the 95UD areas; n = 32) rather than over continental-slope (200–1000 m: 6 ± 2%) or continental-shelf waters (<200 m: <2%). Estimates of spatial overlap among petrels within bimonthly periods varied between years (July–August: 62 ± 4% in 2006, 49 ± 9% in 2007; September–October: 55 ± 7% in 2006, 48 ± 7% in 2007), but were not statistically significant ($P > 0.10$).

Greatest weighted $\Sigma_n$ 95UD values were located primarily between 30–45°S, where we identified three main hotspots of Grey-faced Petrel distribution at sea (fig. 2): (1) the "Ruamaahua hotspot" included the region centred on the breeding colonies on the Ruamaahua Islands; (2) the predominant "Tasman hotspot" was located in the Tasman Sea; and (3) the "Chatham hotspot" was centred on the Chatham Rise. Other smaller hotspots also were detected in the vicinity of the Kermadec–Tonga trenches and Norfolk Island (fig. 2).

The extent and location of the Ruamaahua hotspot displayed spatio-temporal variability. In the July–August period, it extended from the Ruamaahua Islands approximately parallel to the northern coastline of New Zealand's North Island, in a northwesterly direction in 2006 and in an easterly direction in 2007 (fig. 2). The fact that all individuals were observed in the Ruamaahua hotspot during the July–August period in each year reflects departure and arrivals after being captured at the colony (table 4). A lower proportion of individuals (64–93%) was recorded within this hotspot during the September–October periods (fig. 1) because most of the individuals captured in July did not return to the colony during this period (tables 2, 4). The mean percentage area of individual home ranges overlapping the Ruamaahua hotspot ranged between 10 and 18% (fig. 1, table 4).

The location and extent of the Tasman hotspot also displayed spatio-temporal variability. In 2006, the Tasman hotspot extended from approximately 170–155°E and from 30–40°S during the July–August period (fig. 2), but then shifted approximately 5° west during the September–August period. During the latter period, the hotspot paralleled the east coast of Australia from 28–42°S (fig. 2). In 2007, this hotspot was absent in the July–August period (fig. 2), but was present in the September–October period (fig. 2), when it was smaller and centred towards the middle of the Tasman Sea between 155–170°E and 30–38°S. A greater proportion of individuals (86–100%) visited the Tasman hotspot (fig. 1) among all periods, except in July–August 2007 when only 38% of individuals were recorded there (table 4, fig. 2). On average, >50% of each Grey-faced Petrel's total home range area overlapped with the Tasman hotspot, except in July–August 2007 when this was reduced to 13% (table 4).

**TABLE 3**
**Summary of the extent of at-sea distribution data for satellite-tracked Grey-faced Petrels in each bimonthly period in each year.**

| Variable | July–August | | September–October | | Year effect | | | Bimonthly effect | | | Bimonthly × year effect | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 2006 (n = 8) | 2007 (n = 8) | 2006 (n = 14) | 2007 (n = 14) | *F* | d.f. | *P* | *F* | d.f. | *P* | *F* | d.f. | *P* |
| Days tracked | 42.9 ± 2.4 | 36.3 ± 3.5 | 30.9 ± 3.0 | 32.6 ± 3.7 | 0.43 | 1,30 | 0.51 | 13.97 | 1,10 | 0.004 | 9.99 | 1,10 | 0.01 |
| North | 25.6 ± 1.0 °S | 28.1 ± 1.6°S | 27.6 ± 0.9°S | 28.6 ± 0.7°S | | | | | | | | | |
| South | 41.3 ± 1.1°S | 44.2 ± 0.9°S | 41.5 ± 1.0°S | 39.9 ± 1.0°S | 0.003 | 1,30 | 0.96 | 1.65 | 1,9 | 0.23 | 6.92 | 1,9 | 0.03 |
| East | 154.9 ± 1.2°W | 167.7 ± 3.2°W | 154.7 ± 2.1°W | 159.3 ± 2.4°W | 3.67 | 1,30 | 0.06 | | | | | | |
| West | 175.8 ± 3.6°E | 166.1 ± 3.7°E | 177.8 ± 5.2°E | 170.6 ± 4.8°E | 8.01 | 1,30 | 0.008 | 12.77 | 1,10 | 0.005 | | | |

All Grey-faced Petrels were originally captured on the Ruamaahua Islands' breeding colonies (see table 1 for details). Linear mixed effects models were used to test for temporal variation, the Minimum Adequate Models are presented for those where $P < 0.1$.

**TABLE 4**
**Percentage of individuals and the average percentage of individual total 95UD areas which overlapped with three different hotspots (Ruamaahua, Tasman and Chathams) versus other oceanic areas for each bimonthly-year period**

| Variable | Oceanic area | July–August | | September–October | |
|---|---|---|---|---|---|
| | | 2006 (n = 8) | 2007 (n = 8) | 2006 (n = 14) | 2007 (n = 14) |
| Percentage of tagged birds visited area | Ruamaahua hotspot | 100 | 100 | 64 | 93 |
| | Tasman hotspot | 100 | 38 | 93 | 86 |
| | Chathams hotspot | 0 | 88 | 21 | 14 |
| | Other areas | 100 | 100 | 79 | 79 |
| Percentage of home range area | Ruamaahua hotspot | 14 ± 3 | 15 ± 4 | 10 ± 2 | 18 ± 4 |
| | Tasman hotspot | 55 ± 7 | 13 ± 8 | 61 ± 9 | 55 ± 8 |
| | Chathams hotspot | 0 ± 0 | 28 ± 9 | 3 ± 2 | 2 ± 1 |
| | Other areas | 31 ± 7 | 45 ± 9 | 26 ± 7 | 25 ± 8 |

All Grey-faced Petrels were originally captured on the Ruamaahua Islands' breeding colonies (see table 1 for details).

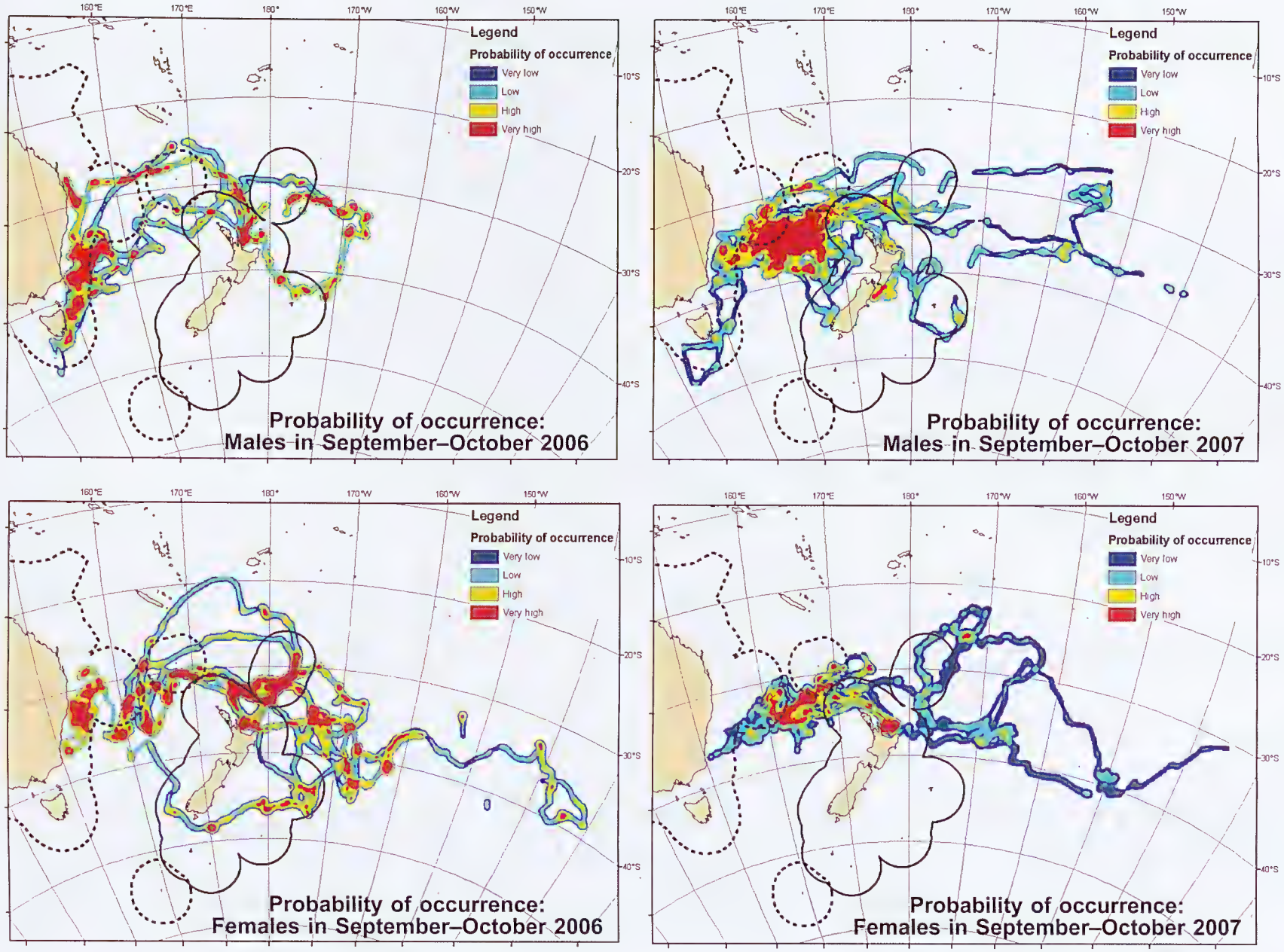


*FIG. 3 — At-sea distribution of satellite-tracked Grey-faced Petrels from the Ruamaahua Islands, New Zealand for males and females in September–October in 2006 and 2007. Maps show the probability of occurrence (2006: $n_{males}$ = 4, $n_{females}$ = 4, 2007: $n_{males}$ = 7, $n_{females}$ = 7). Solid black line indicates the New Zealand Exclusive Economic Zone and the dashed black line indicates the Australian Exclusive Economic Zone.*

The Chathams hotspot only was detected in July–August 2007, when 88% of Grey-faced Petrel individuals travelled to this region. In both years, a small proportion of individuals (14–21%) visited the Chathams hotspot in the September–October period (table 4, figs 1, 2). On average, approximately 28% of each Grey-faced Petrel's total home range area overlapped with the Chathams hotspot in July–August 2007, compared with <4% for all other periods (table 4, figs 1, 2).

Only a small proportion of individuals visited both Tasman and Chatham hotspots (July–August 2006, 0%; July–August 2007, 25%; September–October 2006 & 2007, 14%).

A comparison of the distribution of males and females during the September–October periods (fig. 3) shows both males and females congregated over the Tasman hotspot in both years, with males tending to travel farther west and south than females. In contrast, females tended to travel farther to the east and north of the Ruamaahua Islands in both years, but particularly in 2006, when hotspots of distribution were observed between the Kermedec Trench and the Chatham Rise.

### Use of Exclusive Economic Zones and High Seas by Grey-faced Petrels

Within each bimonthly-year period, the home ranges among all tracked petrels overlapped the High Seas and New Zealand EEZ, except in September–October 2006 when only 71% of home ranges overlapped with New Zealand's EEZ (table 5). The proportions of home ranges that overlapped the Australian EEZ were more variable, ranging from 38% in July–August 2007 to 100% in July–August 2006 (table 5). The average proportion of an individual's 95UD area which overlapped with each zone also varied (High Seas: 38–52%; New Zealand EEZ: 20–49%; Australia EEZ: 5–46%) during the same four bimonthly-year periods (table 5).

After taking into account the relative area available to individuals within each zone, CA revealed no evidence indicating differential use of particular zones during the bimonthly-year periods considered, except in July–August 2006 (table 5). Grey-faced Petrel 95UD areas tended to overlap disproportionately more than expected with the Australian EEZ and less than expected with High Seas during July–August 2006 only. Because zero replacement data can increase the likelihood of a Type-I error in CA (Bingham & Brennan 2004), we evaluated *P* values and the number of zero replacements for each bimonthly-year comparison. The average percentage of individuals with zero utilization values per zone in any given bimonthly-year period was <30% (table 5), and for the only comparison with a significant *P*-value (July–August 2006), there were no zero replacements.

## DISCUSSION

### Satellite tracking

Most satellite telemetry studies investigating the movements of far-ranging pelagic seabirds have involved large-bodied seabirds (>1000 g), with the vast majority of studies targeting adult breeding birds (BirdLife International 2004). As PTTs become smaller and new attachment methods are developed, we anticipate an increase in the use of this technology applied to the much smaller (<1000 g) seabirds including other gadfly petrels (Burger & Shaffer 2008). Although Phillips *et al.* (2003) recognise that PTT deployments on petrels and albatrosses can increase nest desertion and increase foraging trip durations, they suggest that tracking data derived from birds carrying PTTs remains representative of species' foraging areas (see also Ackerman *et al.* 2004, and Wilson *et al.* 2002). We acknowledge that PTTs used in this study (~5% adult body mass) are above the recommended ~3% value suggested by Phillips *et al.* (2003). We do not have the required data to determine if nest desertion among incubating petrels (n = 8 captured in July 2007) observed in this study resulted from handling during the incubation period, detrimental effects of PTTs, or a combination of both factors. Although at least 16% of the 32 tagged petrels returned to the colony at least once, most of which were captured in July during incubation, normal colony attendance patterns for individual breeders on the Ruamaahua Islands during the breeding season remains poorly understood (but see Imber (1976) for information gathered from burrow surveys on Moutohora Island). Due to breeding failures and the tagging of non-/failed breeding individuals, the 2006 and 2007 tracks primarily quantify the movements of individuals free to roam with fewer constraints imposed by the need to return to their colony (as might be expected among adults provisioning chicks at regular intervals). Yet, the degree to which movements of the tagged petrels were affected by the attached tags remains unknown. Provided there are no techno-environmental constraints (i.e., functionality of lightweight solar-powered PTTs deployed in regions with greater cloud cover), future satellite telemetry studies investigating gadfly petrel movements should aim for deployments involving smaller transmitters whenever possible.

### At-sea distribution

Our results indicate that Grey-faced Petrels from the Ruamaahua Islands are confined to the southwestern portion of South Pacific subtropical gyre and the Tasman Sea. This is the first study to describe the at-sea distribution of non-breeding (including failed incubating) Grey-faced Petrels during the breeding season (July–December). Although the movements of the petrels may have been affected by using PTTs, our results are consistent with previous at-sea observations from ships that indicate that the Grey-faced Petrel and its conspecific, *P. m. macroptera*, are restricted to the southern subtropical gyres of the Pacific, Indian and Atlantic oceans (Sladden & Falla 1928, Fleming 1950, Imber 1973). Historical accounts report Grey-faced Petrels were common about 200 miles from the coast of New Zealand to the north and east (Sladden & Falla 1928) and were regularly seen northwards to the Kermadec Islands (30°S; Imber 1973). Our satellite tracking data show that individuals travelled predominantly towards the west of New Zealand into the Tasman Sea. The opposite was reported by Fleming (1950) who suggested that during the breeding season, Grey-faced Petrels range mainly to the east of New Zealand between 31° and 42°S and about 150 km off the coast to 3200 km east (145°W). Based on observations at Moutohora (Whale Island), including stomach contents of chicks, Imber (1973) deduced that breeding Grey-faced Petrels probably forage primarily beyond the continental shelf and, similar to Fleming (1950), he suggested that individuals foraged at least 500 km southeastwards towards the Subtropical Convergence. Imber (1973) also suggested that non-breeders are not likely to be

**TABLE 5**
**Bimonthly-year estimates of the percentage of individuals with available area and home ranges (95% UD areas) that overlapped each zone and the mean (± SE) percentage of areas available and 95UD areas within New Zealand and Australian Exclusive Economic Zones (EEZ) and High Seas for Grey-faced Petrels from the Ruamaahua Islands' breeding colonies**

| Variable | Zone | July–August | | September–October | |
|---|---|---|---|---|---|
| | | 2006 (n = 8) | 2007 (n = 8) | 2006 (n = 14) | 2007 (n = 14) |
| Percentage of individuals with available areas in region | High Seas | 100 | 100 | 100 | 100 |
| | New Zealand EEZ | 100 | 100 | 86 | 100 |
| | Australia EEZ | 100 | 100 | 100 | 100 |
| Percentage of home ranges overlapping each region | High Seas | 100 | 100 | 100 | 100 |
| | New Zealand EEZ | 100 | 100 | 71 | 100 |
| | Australia EEZ | 100 | 38 | 93 | 86 |
| Percentage of total available areas | High Seas | 69 ± 1 | 67 ± 3 | 64 ± 4 | 70 ± 3 |
| (Mean ± SE) | New Zealand EEZ | 24 ± 2 | 26 ± 4 | 15 ± 3 | 21 ± 3 |
| | Australia EEZ | 8 ± 1 | 7 ± 1 | 21 ± 6 | 9 ± 3 |
| Percentage of total home range area | High Seas | 52 ± 5 | 46 ± 9 | 38 ± 5 | 48 ± 6 |
| (Mean ± SE) | New Zealand EEZ | 20 ± 3 | 49 ± 10 | 17 ± 5 | 27 ± 5 |
| | Australia EEZ | 28 ± 5 | 5 ± 2 | 46 ± 8 | 25 ± 5 |
| Rank[1] | High Seas | 3 | 2 | 2 | 3 |
| | New Zealand EEZ | 2 | 1 | 3 | 2 |
| | Australia EEZ | 1 | 3 | 1 | 1 |
| Compositional analysis | λ | 0.253 | 0.371 | 0.801 | 0.846 |
| | P | 0.046 | 0.094 | 0.362 | 0.408 |

Compositional analysis was used to test for whether, relative to an individual's available area in each zone, the home ranges overlapped disproportionately with specific zones.
[1] Relative use of zones was ranked where 1 = high use relative to available areas and 3 = low use relative to available areas, based on compositional analysis.

as constrained as breeders by the need to return regularly to their colony, so they can range farther from colonies (but see, Stahl & Sagar (2006) regarding non-breeder movements in an albatross). Currently, it is not known whether petrels affiliated with Moutohora have a different distribution at sea. Consistent with suggestions by Imber (1973), Grey-faced Petrels tracked from the Ruamaahua Islands occurred predominantly (>90% of time at sea) over deep (>1000 m) offshore waters. This affiliation for pelagic waters also is consistent with observations from summer (January) surveys in the southern Indian Ocean, where *P. m. macroptera* dominated the deep-water offshore avifauna north of the Subtropical Convergence between ~30 and 38°S, albeit at low densities (<1 bird km$^{-2}$; Hyrenbach *et al.* 2007). Together these results indicate that Grey-faced Petrel conspecifics are probably affiliated with subtropical pelagic areas throughout the year.

## Oceanographic setting structuring Grey-faced Petrel distributions

At the mega-scale (i.e., the entire breeding season range among non-/failed breeders), sea surface temperatures range from 4–28°C in the region occupied by Grey-faced Petrels (Ridgway *et al.* 2002, MacLeod *et al.* unpubl. data). Winter (July through September) productivity (mean surface chlorophyll concentrations), especially northeast of New Zealand and in the north-central Tasman Sea, is greater than in surrounding waters (Gregg & Conkright 2001, MacLeod *et al.* unpubl. data). Elsewhere, oceanic boundaries and fronts also have been shown to influence the distribution and abundance of seabirds. For example, Spear *et al.* (2001) described the importance of temperature gradients and vertical hydrography in structuring pelagic seabird communities in the tropical Pacific. They show that piscivorous seabird densities (e.g., Juan Fernandez Petrel *Pterodroma externa* (Salvin, 1875)) were more strongly influenced by the depth and intensity of the thermocline than the location of the Equatorial Front. They suggest that avian piscivore densities were greatest where the thermocline was deepest and most stratified, because this is where their prey were presumably most concentrated.

Individual petrels tracked from the Ruamaahua Islands aggregated within areas defined by well-described, semi-permanent oceanographic circulation features, including (1) the Subtropical Front (STF) centred over the Chatham Rise, (2) the East Auckland Current, (3) the Tasman Front, and (4) the East Australian Current (EAC; Chiswell &

Rickard 2006, Roemmich & Sutton 1998, Ridgway *et al.* 2002). The locations of these features are constrained by the underlying basin, ridge and plateau bathymetry (Chiswell & Rickard 2006). Furthermore, the EAC separation zone (i.e., the region where the EAC splits into two parts: the Tasman Outflow and the Tasman Front) is characterised by large-scale turbulence, which spawns numerous cyclonic and anti-cyclonic eddies, the largest and strongest of which occur near the coast, south of 32° (Nilsson & Cresswell 1981, Mata *et al.* 2006). Although the importance of these features to Grey-faced Petrels remains unknown, we currently are examining how eddy fields are associated with strong physical and chemical gradients known to enhance biological production of potential micronekton (Lutjeharms *et al.* 1985) and cephalopod (Lansdell & Young 2007) prey. These features may indeed be important habitat for foraging Grey-Faced Petrels.

## Spatial overlap and hotspots of distribution at sea

The degree to which the kernel-based UDs within individual's home ranges (i.e., 95UD areas) overlap provides unique information about space-use-sharing throughout the sample population's large range at sea. We found a seemingly high degree of overlap (mean range: 48–62%) among Grey-faced Petrels' home ranges. Somewhat contrary to our finding, observations of Grey-faced Petrels at sea indicate they rarely congregate in groups, except when resting on the sea surface during the day (Imber 1973, Hyrenbach *et al.* 2006). Variation in spatial overlap, however, is currently not well understood but could reflect individual variability in searching strategies and foraging destinations (i.e., including breeders vs. non-breeders), inter-annual differences in weather or oceanographic conditions, or changes in the distribution or availability of key prey resources. These data indicate that among Grey-faced Petrels attending the breeding colonies on the Ruamaahua Islands, individuals appear to occupy similar areas within their vast range at sea during the Austral winter and spring. Furthermore, the tendency for individuals to overlap within certain areas at sea, allowed us to better identify several key hotspot areas.

Our study identified three hotspots for Grey-faced Petrel distribution at sea: the Ruamaahua, Tasman and Chatham hotspots. The Ruamaahua hotspot was centred, as expected, on the Ruamaahua Islands, the location of the breeding colonies and the deployment site for the tracked petrels. This hotspot appears to encompass the predominant flight paths used by the petrels heading away from the colonies along the coast of the North Island of New Zealand. The slight shift in proximity of this hotspot between years may reflect differences in the wind and oceanic conditions in each season. The extent and location of the Tasman hotspot also varied between years and bimonthly periods, which may be related to temporal shifts in the location and intensity of the EAC and the associated Tasman Front. Expected inter-annual variability in the oceanography and local variations in marine productivity may help explain the shift in the extent and location of the Tasman hotspot, and perhaps the emergence of the Chatham hotspot in 2007. There also exists the possibility that the Chatham hotspot identified in July–August 2007, resulted from differences in the movements and subsequent distribution among breeding petrels captured within nesting chambers. This indicates that breeders and non-breeders may occupy different regions at sea, potentially to avoid interference competition. A similar pattern has recently been observed among non-/failed breeding vs chick-provisioning Hawaiian Petrels outfitted with PTTs (J. Adams & D. G. Ainley unpubl. data). A more detailed investigation that incorporates oceanography and satellite-derived locations among Grey-faced Petrels will provide a better understanding of the importance of oceanographic features that influence the location of the hotspots areas.

The location of these hotspots also may be influenced by key geographic features in the cooler, more productive waters of the Tasman Sea, where predatory fish species such as tunas (e.g., *Thunnus maccoyii* Castelnau, 1872) and swordfishes (e.g., *Xiphias gladius* Linnaeus, 1758) are known to feed (Young *et al.* 1996, 2006). Grey-faced Petrels are thought to rely, in part, on these subsurface predators to locate and flush prey to the ocean surface. Swordfish, for example, feed over the Tasman Seamount and Lord Howe Rise, which are flushed by the colder, more productive subantarctic waters and support increased prey biomass (Campbell 2002, Young 2004).

Documentation of at-sea hotspots for Grey-faced Petrel distribution in the Tasman Sea and near the Chatham Islands has important implications in the wake of potential climate change impacts to this region. Climate modelling indicates that changes in the Southern Annular Mode (SAM), the dominant mode of variability in the southern hemisphere atmospheric circulation, could cause dramatic climate-induced change especially to the EAC, but also in the southwestern Pacific (Cai *et al.* 2005). Predicted changes in the near future (50–100 yrs) include an increase in wind-stress-curl east of the Chatham Islands, and warming and strengthening of the EAC (Cai *et al.* 2005). Predicting the effects of these changes on Grey-faced Petrel distribution and foraging ranges requires a more detailed understanding of the importance of these different oceanographic and climatic conditions in determining their movement patterns.

## Implications for multinational ocean management

Determining the spatial and temporal patterns in the at-sea distribution of Grey-faced Petrels is important from a management perspective. Effective conservation and management for extremely mobile and far-ranging marine vertebrates depends increasingly on multinational efforts (Lewison *et al.* 2004), especially when target organisms transverse geopolitical boundaries (e.g., BirdLife International 2004). In a first effort to identify nations that oversee marine resources which provide important habitat for Grey-faced Petrels breeding on the Ruamaahua Islands, New Zealand, we measured habitat use among individuals that occupied three geopolitical marine zones: New Zealand's EEZ, Australia's EEZ, and the area within the petrel's range intersecting the High Seas. At least a third of the area within each Grey-faced Petrel's home range overlapped with the High Seas jurisdictional zone, with the remaining area split, to varying degrees, between the New Zealand and Australian EEZs. Whereas New Zealand entities (iwi, the Ministry for Fisheries and the Department of Conservation) oversee protection, conservation and management for Grey-faced Petrels and other seabirds in their jurisdictional region, jurisdiction over the High Seas in the Tasman Sea and southwestern Pacific Ocean is shared by the Western and Central Pacific Fisheries Convention (WCPFC) nations (BirdLife International 2004).

Because gadfly petrels often feed in association with subsurface predators, some species are at risk from certain industrialised fishing practices (such as longlining) that target these fishes. Furthermore, many gadfly petrels scavenge food from the surface, including baited hooks, and other procellariid seabirds potentially are attracted to or disoriented by artificial light sources at sea (Reed *et al.* 1985). Multinational fishing operations targeting pelagic fishes such as tunas and billfishes operate to varying degrees in waters used by Grey-faced Petrels. Although many of these fisheries are increasingly adopting measures to reduce seabird bycatch, observer coverage among many fisheries is insufficient to assess accurately the impact of fisheries activity on procellariiform seabird populations (Lewison *et al.* 2004).

We have made preliminary progress by identifying key use areas for Grey-Faced Petrels. Future studies should aim to investigate the extent of overlap between the at-sea distribution of Grey-faced Petrels (of known breeding status if possible) and commercial fisheries activity to determine the risk that this industry poses for this species. Research should also determine the environmental factors influencing the at-sea distribution of Grey-faced Petrels to provide a better understanding of the potential impact of climate change. Without this information, resource managers would be unable to assess (1) direct and indirect effects of commercial fisheries, or (2) the effect of oceanic and atmospheric climate on Grey-faced Petrel population dynamics (i.e., survival, abundance).

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# MAXIMUM DIVE DEPTHS OF EIGHT NEW ZEALAND PROCELLARIIFORMES, INCLUDING *PTERODROMA* SPECIES

by G. A. Taylor

(with four text-figures, one plate and one table)

**Taylor, G.A.** 2008 (31:x): Maximum dive depths of eight New Zealand Procellariiformes, including *Pterodroma* species. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 89–98. ISSN 0080-4703. Research & Development Group, Department of Conservation, PO Box 10420, Wellington, New Zealand. Email: g.taylor@doc.govt.nz

Lightweight capillary tube depth gauges were attached to eight petrel species breeding at New Zealand colonies during the period 1998–2008. This paper presents the first information on the diving ability of *Pterodroma* petrels. Grey-faced Petrels, *Pterodroma macroptera gouldi*, recorded maximum dives down to 23 m. Males (6.3 ± 6.3 m SD) dived deeper on average than females (3.6 ± 2.5 m) during the incubation period but not significantly so (P=0.06). Breeding birds dived significantly deeper on average than non-breeders, and breeding males dived significantly deeper on average than non-breeding males. The two small *Pterodroma* species sampled, *Pterodroma pycrofti* and *Pterodroma nigripennis*, only exhibited shallow dives down to 2 m but sample sizes were small. Sooty Shearwaters, *Puffinus griseus*, had mean maximum dive depths of 42.7 ± 23.7 m, with males (53.0 ± 17.3 m) diving significantly deeper on average than females (20.1 ± 20.4 m) during the incubation period. One male Sooty Shearwater dived to nearly 93 m, the deepest dive so far recorded in the order Procellariiformes. Flesh-footed Shearwaters, *Puffinus carneipes*, dived to 28 m, with a mean maximum dive depth of 13.6 ± 7.9 m. Hutton's Shearwaters, *Puffinus huttoni*, had a mean maximum dive depth of 23.0 ± 8.5 m (range 11.1–36.6 m). A single Fluttering Shearwater, *Puffinus gavia*, recovered with a dive gauge had dived to 29 m. Mean maximum dives made by Common Diving-Petrels, *Pelecanoides urinatrix*, of 10.9 ± 6.1 m (range 6.9–22.2 m) were shallower than results reported from other sites but may have been biased by gauge failures. Capillary gauges provide the best means we have at present to understand the diving capability of small seabirds. While studies elsewhere have shown these gauges may overestimate diving performance by about 10–15%, other factors identified in this study indicate that sometimes diving performance will be underestimated using this simple technique.

**Key Words: maximum dive depths, sex-specific behaviour, seabirds, *Pterodroma*, *Puffinus*, *Pelecanoides*, New Zealand.**

## INTRODUCTION

Electronic time depth recorders (TDRs) have been used to examine dive profiles of many large and medium-sized seabirds such as King Penguins, *Aptenodytes patagonicus* J.F. Miller, 1778 (Sato *et al.* 2002), Little Penguins, *Eudyptula minor* J.R. Forster, 1781 (Ropert-Coudert *et al.* 2006) and shags (Wanless *et al.* 1999). Early model TDRs were relatively heavy (40–50 g) which limited their usefulness for small seabirds (<1 kg). In recent years, smaller and lighter TDRs (down to16 g) have been used on seabirds as small as Rhinoceros Auklets, *Cerorhinca monocerata* (Pallas, 1811), weighing 550–600 g (Watanuki *et al.* 2006), but these devices are still quite expensive for large-scale deployment. Low-cost and lightweight (<1 g) plastic capillary tube depth gauges (Burger & Wilson 1988) have been attached to seabirds since the 1980s to examine diving performance, especially the maximum depth reached by seabirds. The technique has been used on a wide range of seabird groups, including penguins (Sphenisciformes), gannets, boobies, tropicbirds and cormorants (Pelecaniformes) and alcids (Charadriiformes) (Burger & Simpson 1986, Scolaro & Suburo 1991, Adams & Walter 1993, Le Corre 1997, Casaux *et al.* 2001). Eighteen species of tube-nosed seabirds (Procellariiformes) have had their maximum dive depths recorded (Prince *et al.* 1994, Bocher *et al.* 2000, Brooke 2004); the notable absence in this group includes members of the genus *Pterodroma*.

Maximum depth gauges (MDGs) consist of short lengths of fine gauge (c. 1 mm internal diameter) tubing that are attached to foraging seabirds. MDGs allow dive depths to be estimated through compression of a known volume of air inside the tubes, marked with indicator powder. Seawater drawn into the tubes, as a result of air compression as the bird dives, dissolves this powder leaving a boundary mark from which calculation of dive depth can be made.

In 1998, the New Zealand Department of Conservation began trials on an underwater line-setting device as part of a research program to develop techniques to mitigate the impact of seabird bycatch by long-line fisheries. Prior research on White-chinned Petrels, *Procellaria aequinoctialis* Linnaeus, 1758, (Huin 1994) and albatross (Prince *et al.* 1994) indicated that some Procellariiformes had much greater diving abilities than previously suspected (Warham 1990). Capable diving species were exposed to a greater risk behind fishing boats as they could pursue sinking baits on hooks deeper into the water column. Seabird species at risk from taking baited hooks in the New Zealand region (Robertson *et al.* 2004) included several species with unknown diving ability (Brooke 2004). The early part of this study included an assessment of the diving ability of Grey-faced Petrels, *Pterodroma macroptera gouldi* (Hutton, 1869), and Flesh-footed Shearwaters, *Puffinus carneipes* Gould, 1844, two common breeding species in northern New Zealand and known to be captured by fishing fleets targeting tuna species (Robertson *et al.* 2004).

MDGs were attached to Sooty Shearwaters, *Puffinus griseus* (J.F. Gmelin, 1879), to compare the diving ability of birds nesting in northern New Zealand with those birds breeding at the Snares Islands, 11° of latitude further south (Weimerskirsch & Sagar 1998). Over the following 10 years, further seabird species were sampled on an opportunistic basis during other studies (e.g., Miskelly & Taylor 2004). The five additional species included Common Diving-Petrel, *Pelecanoides urinatrix* (J.F. Gmelin, 1789), Hutton's and Fluttering shearwaters, *Puffinus huttoni* Matthews, 1912, and *P. gavia* (J.R. Forster, 1844), Black-winged and

Pycroft's petrels, *Pterodroma nigripennis* (Rothschild, 1893) and *P. pycrofti* Falla, 1933.

The aim of this paper is to bring together dive depth data across a range of New Zealand Procellariiformes (four main study species and four other species with smaller sample sizes) to compare diving behaviour and its role in niche separation, at an interspecific and intraspecific level. The partitioning of diving behaviour between sexes and birds of different breeding status is also examined. These types of data are also useful in the ongoing development of fisheries mitigation techniques for several at-risk seabird species and will help inform managers about the risks to diving seabirds from proposed new technologies such as tidal power generation plants.

## METHODS

Fieldwork was carried out at seven different seabird colonies around New Zealand during the period 1998–2008. The main study areas were on Kauwahaia and Ihumoana islands, Bethell's Beach, near Auckland (36°54'S, 174°26'E) (pl. 1) between 5 July 1998 and 29 December 1999. Grey-faced Petrels, Sooty Shearwaters and Flesh-footed Shearwaters breed on Kauwahaia Island and Grey-faced Petrels breed on Ihumoana Island. The Hutton's Shearwater breeding colony in the Kowhai River, Seaward Kaikoura Range (42°16'S, 173°36'E), was visited from 11–20 September 2001. Opportunistic sampling of petrels was carried out at the remaining study sites. Motumahanga Island (Sugarloaf Islands), near New Plymouth (39°03'S, 174°01'E) was visited from 16–20 November 1998. Flesh-footed Shearwaters and Common Diving-Petrels breed at this site. North Brothers Island, Cook Strait (41°06'S, 174°25'E) was visited from 25–27 November 1998. Common Diving-Petrels breed at this site. East Island (37°41'S, 178°34'E) was visited from 19–23 November 2001. Fluttering Shearwaters and Black-winged Petrels breed at this site. Cuvier Island (36°25'S, 175°46'E) was visited from 4–11 February 2008. Pycroft's Petrels now breed at this newly established colony (11 pairs in 2008).

MDGs used in this study were made from lengths of clear flexible PVC plastic tubing (2.0 mm external diameter, 0.8 mm internal diameter) supplied by Ceelon Plastics Ltd, Wellington. Variable lengths of tubing were cut (typically 120 mm but lengths ranged between 76 mm and 200 mm depending on the size of the species). Shorter tubes were used on small species such as Common Diving-Petrels. A thin layer of dry icing sugar was drawn through the tubes after slightly moistening the interior by blowing through the tube. One end of the tube was heated and sealed by squeezing together with pliers. However, during sampling on Common Diving-Petrels in November 1998, I found this sealing technique was not always reliable. This problem was avoided in subsequent deployments during 1998 and 1999 by sealing the closed end of the tube with wax and/or epoxy resin. Heated wax or epoxy resin was drawn up slightly into the tube which was then heat-sealed to provide a more secure fix. From 1999 onwards, the closed end was dipped and covered in epoxy resin after heat-sealing.

MDGs were attached to the central tail feathers of most species using waterproof adhesive tape (several brands of ducting tape). Species with very short tail feathers (e.g., Common Diving-Petrels, Fluttering and Hutton's shearwaters) had the MDGs attached by tape to the central back feathers. To avoid problems with water being forced up the tubes if the birds plunge-dived, the open end of the tube was pointed towards the end of the tail. Attachment generally took only 1–3 minutes per bird with MDGs being attached to birds caught on the ground at night or on birds found incubating eggs or feeding chicks. All the birds used in this study were also fitted with metal bands supplied by the New Zealand National Bird Banding Scheme.

*PLATE 1*
*Kauwahaia Island, Bethell's Beach, New Zealand. This island was the main study site for the work on Grey-faced Petrels, Sooty and Flesh-footed shearwaters in 1998 and 1999. A Sooty Shearwater breeding on this island made the deepest dive (93 m) so far recorded in any species of Procellariiformes.*

Birds were recaptured on the ground on subsequent nights or located in nesting chambers after they returned from a foraging trip. Upon recovery of each MDG, the total length of the tube was measured from the opening to the heat sealed end or where the wax or epoxy resin started adjacent to the sugar-coated section. The length of clear tubing was then measured to the nearest 0.5 mm and the sugar-coated portion remaining in the tube was obtained by subtracting the total length of tube from the clear portion lacking sugar. Maximum diving depth was calculated by the equation: $D = 10.08 ([Ls/Ld]-1)$ where D is depth (m), 10.08 is the height (m) of a column of sea water equivalent to 1 atmosphere of pressure, Ls is the initial length (mm) of icing sugar and Ld the length (mm) of icing sugar remaining after the foraging trip (Burger & Wilson 1988).

All of the breeding Grey-faced Petrels, Flesh-footed Shearwaters and Sooty Shearwaters sampled at the Bethell's Beach colonies were sexed by cloacal examination just after laying (data collected 1989–2007). Additionally, every bird sampled in the trials at the Bethell's Beach colonies had their sex validated by DNA tests carried out on blood samples collected between 1996 and 2001. The sex of birds at other study sites used in this paper was unknown.

### Grey-faced Petrels

MDGs were attached to 199 Grey-faced Petrels on Kauwahaia and Ihumoana islands in 1998. The first batch of MDGs was attached to 48 males and 76 females from 5–13 July 1998 (during the early incubation period). Twenty of these birds (three males and 17 females) were recaptured and released again with a replacement MDG in July or August. Two males and one female from the first batch had a third MDG

deployment. Another 45 different birds (15 males, 30 females) were captured from 20–23 August 1998 (during the hatching period) and released with a second batch of MDGs. Six of these birds (three males and three females) were recaptured in August and released with a replacement MDG. One female from the second batch had a third MDG deployment.

Non-breeding Grey-faced Petrels were separated from breeding birds by a combination of behavioural traits and examination of differences on their bodies. All burrows on the two islands were monitored each season so breeding occupants were identified as banded birds incubating eggs or feeding chicks in each burrow. Non-breeding birds were identified by their behaviour during the species incubation period (calling and roaming activity on the ground at night, displaying to other birds away from burrows, responding to human calls or sleeping on the surface). The weights of non-breeding birds were also lighter than breeders, and cloacal examination around the time of laying separated non-breeding females from those that had laid eggs. Brood patch status was checked for all birds during the incubation period. The brood patches of breeding birds were bare with abundant blood vessels on the surface but those of non-breeders were downy or lacking obvious vascular tissue.

### Flesh-footed Shearwaters

MDGs were attached to 12 non-breeding Flesh-footed Shearwaters (sex unknown) on Motumahanga Island from 16–18 November 1998 during the pre-laying period. MDGs were attached to 20 incubating adults (ten males and ten females) on Kauwahaia Island from 1–11 December 1998. Seventeen breeding birds (eight males and nine females) had MDGs attached from 5–11 December 1999; five of these males and two of the females had also been used in the 1998 deployment.

### Sooty Shearwaters

MDGs were attached to 45 incubating Sooty Shearwaters (29 males and 16 females) on Kauwahaia Island from 1–11 December 1998. Thirty incubating Sooty Shearwaters were sampled on the same island from 5–11 December 1999. These included six males and seven females not used in 1998. However, 16 males and one female had MDGs attached previously in 1998.

### Hutton's Shearwaters

MDGs were attached to 29 Hutton's Shearwaters from 11–20 September 2001 during the pre-laying courtship period. At this stage of the breeding season, heavy snow covered the burrow entrances preventing access to nest chambers resulting in birds only visiting the colony overnight before returning to sea.

### Other petrel species

MDGs were attached to 26 Common Diving-Petrels on Motumahanga Island from 16–18 November 1998 and on two birds on North Brothers Island from 25–27 November 1998. As most birds were captured on the ground soon after landing, the sample represents an unknown proportion of breeding and non-breeding birds.

Six Fluttering Shearwaters of unknown status were captured on East Island on 19 November 2001 and had MDGs attached. Birds at this stage of the season were feeding young chicks.

MDGs were attached to 14 Black-winged Petrels on East Island on 19 November 2001 during the pre-laying courtship period. Ten Pycroft's Petrels on Cuvier Island had MDGs attached on 4–5 February 2008. Some were adults feeding chicks and the remaining birds were non-breeders.

### Statistics

The effect of MDG deployment duration on comparisons made between sexes and birds of different breeding status for Grey-faced Petrels, and between sexes of Flesh-footed and Sooty shearwaters, was assessed using an ANCOVA model on JMP 5.1 (SAS Institute Inc), with the days to recovery being the covariate. All data were log transformed prior to running this model. Data normality was assessed using Kolmogorov-Smirnov tests. Log transformations were needed to normalise the skewed Grey-faced Petrel dataset. Differences in the means of normally distributed data were assessed using Student t-tests for differences between sexes and breeding status. Correlations between dive depths and days until MDG recovery were analysed using Pearson correlation coefficients. Maximum dive depths are given as means ± SD. Statistical tests and descriptive statistics were run on the program SigmaStat ver. 3.0., with the exception of the ANCOVA model.

## RESULTS

Mean (± SD), median and range of maximum dive depths for the eight species studied in this project are summarised in table 1. This table also includes the mean (± SD) and range of MDG deployment duration (the number of days between placement of MDG on the birds and their subsequent recapture) for each species. The ANCOVA models found no effect of deployment duration on the differences obtained between sexes or breeding status of birds in the Grey-faced Petrel, Flesh-footed Shearwater or Sooty Shearwater datasets.

### Grey-faced Petrel

Only 102 (51%) of the Grey-faced Petrels deployed with MDGs were recaptured again in July or August 1998. Thirty-seven (36%) of these birds had MDGs that malfunctioned or were lost. Eighteen had puncture holes in the tubes or the heat seals had broken, as a result of the birds attempting to remove the tubes. Twelve birds returned with no MDGs because they had lost tail feathers or the tubes detached from the tape. A further seven birds were recovered with a dirt plug blocking the end of the tube and the icing sugar still intact. No dive depth data were recorded for these 37 individuals. MDGs with readable sugar traces were recovered from 65 birds (64%), including 10 birds that were sampled twice and one bird that was sampled three times. To avoid pseudo-replication issues from non-independence of samples, only the maximum dive depth recorded for each individual is used in the analysis for this species.

The mean maximum diving depth reached by sampled Grey-faced Petrels (n=53) was 4.7 ± 4.7 m (range 0.7–23.6 m) (table 1). Maximum dives by both sexes did not produce normal distributions (K-S test 0.22 for females, 0.27 for males, $P<0.0001$) (fig. 1) and a log transformation was used to normalise these data before conducting the

Student t-test analysis. The mean maximum dive depths of males (6.3 ± 6.3 m, n = 22) and females (3.6 ± 2.5 m, n = 31) were not significantly different (t = 1.92, df = 51, P = 0.06) (fig. 1).

Breeding Grey-faced Petrels (n = 25) dived significantly deeper than non-breeding birds (n = 28) (6.6 ± 5.8 m versus 3.0 ± 2.3 m; t = 3.6, df = 51, P <0.01) Breeding males (n = 17) dived significantly deeper than non-breeding males (n = 5) (7.5 ± 6.7 m versus 2.2 ± 0.9 m; t = 2.31, df = 20, P = 0.03). However, the dives of breeding females (4.6 ± 2.4 m, n = 8) were not significantly deeper than those of non-breeding females (3.2 ± 2.5 m, n = 23) (t = 1.85, df = 29, P = 0.07). There was a highly significant relationship between MDG deployment duration and diving depth (r=0.52, df = 51, P <0.01). However, apart from two MDGs that gave much deeper readings after the longer deployment period, the remaining MDGs had a similar distribution of dive depths when deployed between 1 and 10 days compared with those deployed for 38–54 days (fig. 2).

## Flesh-footed Shearwater

Five (42%) of the Flesh-footed Shearwaters sampled on Motumahanga Island were recaptured with readable MDGs. Eighteen (90%) of the Flesh-footed Shearwaters sampled on Kauwahaia Island in 1998 were recaptured between 6 December 1998 and 2 January 1999. Four of these birds were recaptured twice after a subsequent set of MDGs was deployed. Ten (59%) of the birds sampled in 1999 on Kauwahaia Island were recaptured from 10–29 December 1999. Readable MDGs were present on 22 birds (15 in 1998 and seven in 1999). These included four individuals that had two maximum dive depth samples collected. To avoid pseudo-replication issues from non-independence of samples, only the maximum dive depth recorded for each individual (n=18) recorded over the two sampling years on Kauwahaia Island is used in the analysis for this species.

The mean maximum dive depth for all sites and years combined was 13.6 ± 7.9 m (range 0.8–28.7 m, n = 23) (table 1). The mean maximum dive depth recorded from the incubating birds at Kauwahaia Island was 16.8 ± 5.5 m (range 8.2–28.7 m, n = 18). There was no significant difference (t = 1.94, df = 16, P = 0.07) in the dives of males from Kauwahaia Island (18.7 ± 5.2 m, n = 11) compared with females (13.9 ± 5.1 m, n = 7). There was a significant positive relationship between the duration that MDGs were on birds and maximum diving depth (r = 0.45, df = 21, P = 0.03), (fig. 3). However, there was no relationship between deployment duration and maximum dive depths when only the incubating birds at Kauwahaia Island were tested (r = –0.16, df = 16, P = 0.52).

## Sooty Shearwater

Forty-four (59%) of the Sooty Shearwaters deployed with MDGs in 1998 and 1999 were subsequently recaptured within 30 days. Ten (22%) of the birds sampled in the 1998 season and nine (30%) sampled in the 1999 season were recaptured with readable MDGs. These included three individuals that had two maximum dive depth samples collected. To avoid pseudo-replication issues from non-independence of samples, only the maximum dive depth recorded for each individual (n=16) is used in the analysis for this species.

The mean maximum dive depth for the sampled Sooty Shearwaters (both sexes) for 1998 and 1999 combined was 42.7 ± 23.7 m (range 1.2–92.9 m, n=16) (table 1). Males (53.0 ± 17,3 m, n=11) dived significantly deeper than females (20.1 ± 20.4 m, n=5) (t =3.33, df =14, P<0.01). Dive depths increased significantly with the duration that MDGs were deployed on birds (r =0.54, P=0.03, n=16), (fig. 4).

One male Sooty Shearwater (Z-20733) on Kauwahaia Island produced a maximum dive of nearly 65 m after only two days at sea. However, this dive is not included in the results above as this same bird went on to register a dive of almost 93 m on the next MDG deployment, the deepest dive so far recorded for this species.

## Hutton's Shearwaters

Fifteen (52%) of the Hutton's Shearwaters were recaptured at the breeding colony up to five days after MDG deployment. Thirteen (45%) of these birds had intact, readable MDGs and the other two had lost their tubes but retained the tape on the feathers. Hutton's Shearwaters had a mean maximum dive depth of 23.0 ± 8.5 m (range 11.1–36.6 m, n = 13) (table 1).

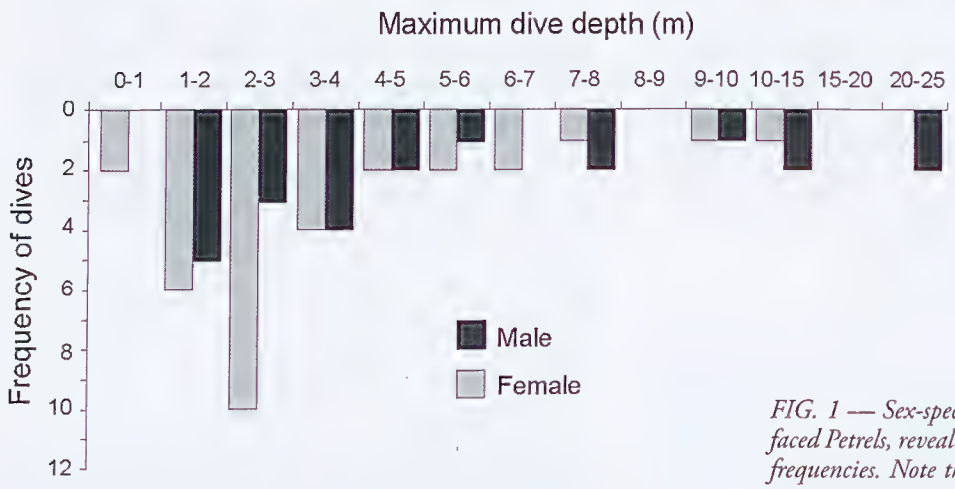


*FIG. 1 — Sex-specific diving behaviour in Grey-faced Petrels, revealed by maximum dive depth (m) frequencies. Note the change in scale over 10 m.*

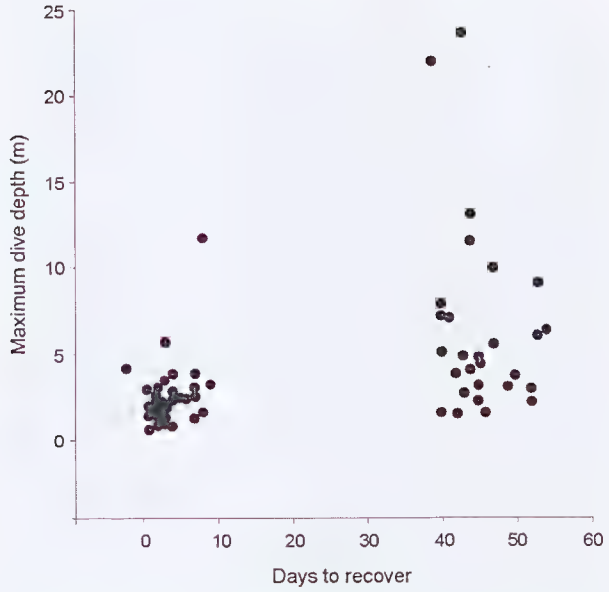


*FIG. 2 — Duration of MDG deployment versus maximum dive depth (m) for Grey-faced Petrels.*

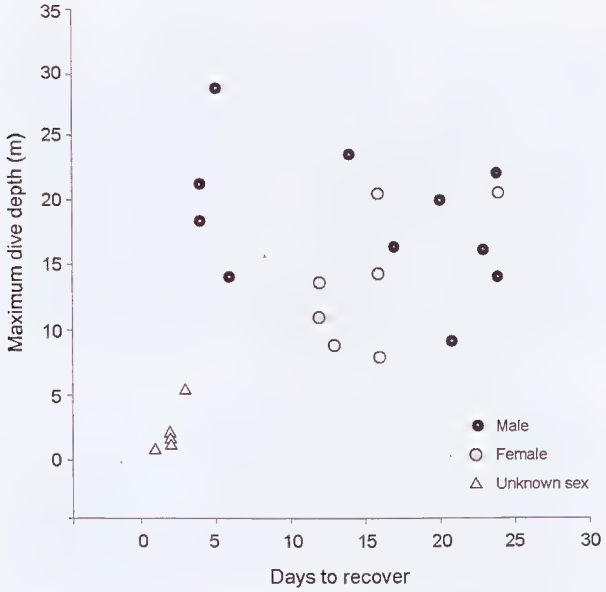


*FIG. 3 — Duration of MDG deployment versus maximum dive depth (m) for Flesh-footed Shearwaters.*

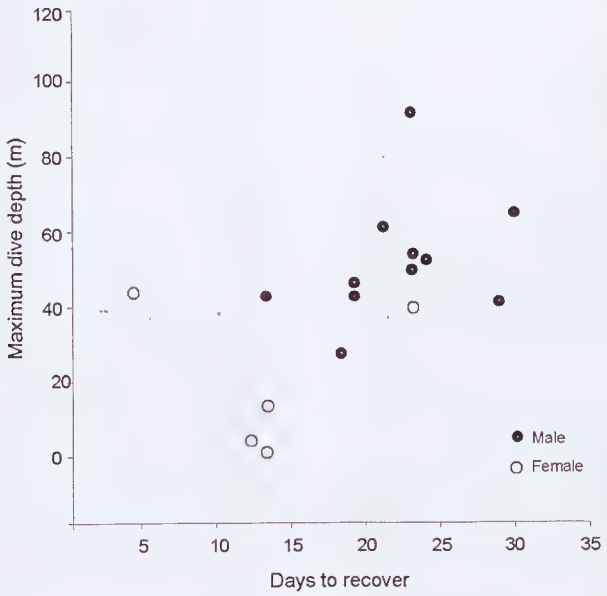


*FIG. 4 — Duration of MDG deployment versus maximum dive depth (m) for Sooty Shearwaters.*

**TABLE 1**
**Summary statistics of the maximum dive depths and trip duration for eight species of petrel**

| Species | Maximum Dive Depths (m) | | | | | Trip Duration (days) | | |
|---|---|---|---|---|---|---|---|---|
| | n | Median | Mean | SD | Range | Mean | SD | Range |
| Grey-faced Petrel | 53 | 3.1 | 4.7 | 4.7 | 0.7–23.6 | 25.9 | 21.0 | 1–54 |
| Black-winged Petrel | 4 | 1.6 | 1.6 | 0.3 | 1.3–1.9 | 2.8 | 0.5 | 2–3 |
| Pycroft's Petrel | 3 | 1.1 | 1.1 | 0.5 | 0.6–1.6 | 2.3 | 0.6 | 2–3 |
| Sooty Shearwater | 16 | 44.3 | 42.7 | 23.7 | 1.2 – 92.9 | 19.3 | 6.9 | 4–30 |
| Flesh-footed Shearwater | 23 | 14.1 | 13.6 | 7.9 | 0.8 – 28.7 | 12.2 | 8.2 | 1–24 |
| Hutton's Shearwater | 13 | 20.3 | 23.0 | 8.5 | 11.1–36.6 | 2.3 | 1.1 | 1–5 |
| Fluttering Shearwater | 1 | 29.4 | – | – | 29.4 | 2 | – | 2 |
| Common Diving-Petrel | 6 | 7.8 | 10.9 | 6.1 | 6.9–22.2 | 1.5 | 0.5 | 1–2 |

### Other petrel species

Eighteen (64%) of the 28 Common Diving-Petrels sampled in 1998 were recaptured one or two nights after MDG deployment. There was a high failure rate for MDGs deployed on this species. Nine (50%) of the MDGs recovered had broken heat seals and no sugar inside the tubes, and three birds had lost their MDG but retained the tape on their feathers. Only six of the birds recaptured (33%) had readable MDGs. Mean maximum dive depth of Common Diving-Petrels sampled at Motumahanga Island (n = 5) and North Brothers Island (n = 1) was 10.9 ± 6.1 m (range 6.9–22.2 m, n = 6) (table 1).

Only one (17%) of the Fluttering Shearwaters deployed with a MDG on East Island was recaptured. This bird had dived 29 m in the two days it was out at sea.

Four (29%) of the Black-winged Petrels were recaptured on East Island 2–3 nights after deployment; all with intact MDGs. These birds had a mean maximum dive depth of 1.6 ± 0.6 m (range 1.3–1.9 m n = 4) (table 1). Three (25%) of the Pycroft's Petrels with MDGs attached were recaptured at the Cuvier Island colony prior to departure from the island on 12 February; all three had readable MDGs. These birds had a mean maximum dive depth of 1.1 ± 0.5 m (range 0.6–1.6, n = 3) (table 1).

## DISCUSSION

### Interspecific comparisons

The maximum dive depths of Grey-faced Petrels, Flesh-footed, Hutton's and Sooty shearwaters (the four species with more than 10 readable MDGs recovered in this study) suggests niche separation with each species using a different zone of water near the ocean surface. Grey-faced Petrels exploit a diverse range of squid species and lesser amounts of fish and crustaceans (Imber 1973). The findings in this study indicate that most of these prey items are gathered near the ocean surface (in the top 5 m of the water column) but some prey could be taken at depths down to 23 m. Shearwaters feed mainly on small fish or crustaceans (especially krill and amphipods) caught during diurnal feeding sessions (Marchant & Higgins 1990, Warham 1990, Cruz *et al.* 2001). Flesh-footed Shearwaters feed mainly on small fish (G. Taylor unpubl. data). This species dived deeper on average than Grey-faced Petrels although their overall depth range was similar. Measurements obtained from MDGs a few days after deployment showed that most dives were less than 5 m deep. However, Flesh-footed Shearwaters foraging during their partner's incubation shift of around 10 days (G. Taylor unpubl. data) routinely dive between 10–30 m. Hutton's Shearwaters feed mainly on small fish and crustaceans (especially krill) (West & Imber 1985). This species is an even more proficient diver than the two previous species and regularly dives between 10–35 m, even during short foraging trips (1–2 days) away from the colony. Sooty Shearwaters feed mainly on small crustaceans (krill and amphipods) but also take squid and fish (Cruz *et al.* 2001). Sooty Shearwaters are the most capable diving species of the four larger petrels examined in this study and short trips away from the colony (1–5 days) regularly resulted in maximum dive depths of 40–60 m and some birds may dive as deep at 93 m.

Brandhorst *et al.* (1971) (cited by Scolaro & Suburo 1991) indicated that anchovies, *Engraulis anchoita* Hubbs & Marini, 1935, around South America formed compact fish shoals at depths greater than 35 m during the day (sometimes down to 75–100 m), while at night these fish dispersed thinly 3–20 m below the sea surface. If the small fish (and other prey species) sought by shearwaters in New Zealand behave in a similar way, then this might explain the deep pursuit diving by shearwaters feeding by day, whereas Grey-faced Petrels which feed mainly at night (Imber 1973) could catch these fish stocks near the surface with shallower dives.

The results of this study suggest that petrels of the genus *Pterodroma* are able to exploit deeper depths in the water column than was previously thought. For example Warham (1990) considered *Pterodroma* species to be mainly surface feeders and Imber (1973) stated that Grey-faced Petrels would probably capture their prey items within 1 m of the sea surface, based on diet samples regurgitated from chicks. However, the measurements obtained from this study show that Grey-faced, Pycroft's and Black-winged petrels routinely dive between 0.5 m and 2 m below the sea surface on foraging trips, and the larger Grey-faced Petrel often dives between 2–10 m, and a few individuals descend as deep as 23 m. Compared with the shearwaters in this study, the majority of *Pterodroma* species sampled made relatively shallow dives (less than 10 m) which is similar to the diving behaviour exhibited by Bulwer's Petrel, *Bulweria bulwerii* (Jardine & Selby, 1828), (maximum depth 5.3 m) (Mougin & Mougin 2000), Blue Petrels, *Halobaena caerulea* (J.F. Gmelin, 1789), (maximum depth 6.2 m) and Slender-billed Prions, *Pachyptila belcheri* (Matthews, 1912), (maximum depth 7.5 m) (Chastel & Bried 1996).

Brown *et al.* (1978) considered that shearwaters might be able to dive down to 5 m. Anecdotal observations by Wood (1993) and Oka (1994) of Flesh-footed Shearwater and Skira (1979) of Short-tailed Shearwater, *Puffinus tenuirostris* (Temminck, 1835), foraging behaviour indicated these species had a diving ability of at least 13 m and 20 m respectively. The diving skills of shearwaters has become much better known in recent years (Weimerskirch & Sagar 1996, Burger 2001) and some species are now reported to dive as deep as 71 m (Weimerskirch & Cherel 1998). This study has shown that Flesh-footed Shearwaters are more proficient divers than previously thought and that the species regularly dives to 10–20 m (perhaps as deep as 30 m). Hutton's and Fluttering shearwaters appear to be capable of routinely diving to 30–35 m.

The diving ability of Sooty Shearwaters during the chick-rearing period was examined using MDGs at the Snares Islands in February and March 1995 (Weimerskirch & Sagar 1996). Their study birds averaged maximum dives of 38.7 m, with dives ranging between 2–67 m. The exceptional diving ability of Sooty Shearwaters was confirmed in the current study in which mean maximum dives (42.7 m and 38.7 m respectively) were very similar for birds breeding at two localities separated by 11° of latitude (one colony breeding in temperate seas, the other in cool sub-Antarctic seas). The maximum dive recorded at The Snares and Kauwahaia Islands was 67 m and 93 m respectively, the latter being the deepest dive recorded for any Procellariiformes species to date.

The maximum dive depths observed in the small sample measured from Common Diving-Petrels in this study (6.9–22.2 m, n=6) were shallower than those reported by Bocher *et al.* (2000) for the same species gathered using MDGs at the Kerguelen Islands (7.9–64.1 m, n=347). They

were also shallower than those reported by Prince & Jones (1992) using MDGs with South Georgian Diving-Petrels, *Pelecanoides georgicus* Murphy & Harper, 1916, at South Georgia (17–48 m, n=6), and Zavalaga & Jahncke (1997) using MDGs with Peruvian Diving-Petrels, *Pelecanoides garnotii* Lesson, 1828, in Peru (10–83 m, n=22). Diving petrels have a remarkable diving ability for their size, but dive depths may be dependent on local food sources and also the depth of the sea floor near the colonies, especially as the species stays quite close to the colonies during chick-rearing (when both parents feed the chick each night) (Miskelly & Taylor 2004). Some deeper dives may have been missed in the current study however because of the high MDG failure rate. The heat seals on the MDGs used with this species were not secured with epoxy resin. Further studies are clearly required on this species in the New Zealand region.

## Sex-specific behaviour

Sex-specific diving behaviour in monomorphic seabirds has been noted previously in several seabird groups (Lewis *et al.* 2002). Using activity loggers, Lewis *et al.* (2002) found that female Northern Gannets, *Morus bassanus* (Linnaeus, 1758), made longer and deeper dives than males. Similarly, a study using MDGs on Antarctic Shags, *Phalacrocorax bransfieldensis* Murphy, 1936, recorded females diving significantly deeper than males and females achieved the maximum dive depth for this species of 112.6 m (Casaux *et al.* 2001). Studies on inter-sexual differences in diving behaviour are limited for Procellariiformes. Mougin & Mougin (2000) failed to detect sex-specific differences in the diving ability of Bulwer's Petrels. However, Peck & Congdon (2006) found that male Wedge-tailed Shearwaters, *Puffinus pacificus* (J.F. Gmelin, 1789), dived significantly deeper on average than females. The findings of this study are similar to those of Peck & Congdon (2006) with male Sooty Shearwaters diving significantly deeper than females during the breeding season. The males of both Grey-faced Petrels and Flesh-footed Shearwaters also dived deeper on average than females but not significantly so. Males in most Procellariiformes are marginally larger than females (Warham 1990) but the difference doesn't seem enough to give a significant advantage during foraging. Peck and Congdon (2006) concluded that inter-sexual competition at the foraging grounds was the most likely explanation for these diving differences.

## Effects of breeding status on diving behaviour

The effects of breeding status on diving behaviour have been studied in cormorants and penguins (Green *et al.* 2005, Grémillet *et al.* 2005). However, this study is, to my knowledge, the first to report differences in diving performance based on breeding status in Procellariiformes. Breeding Grey-faced Petrels (breeding males and for both sexes combined) dived deeper than their non-breeding counterparts. A likely explanation for this phenomenon is that breeding birds may need to obtain different prey items (higher dietary quality or more abundant prey) to restore fat reserves quickly while at sea during their partner's incubation shift. Grey-faced Petrel adults lose over 20% of their body weight during long incubation shifts. For males in particular, who undertake two long incubation shifts of 2–3 weeks each over a two-month period (Imber 1976), rapid recovery of body condition is essential for breeding success (Brooke 2004). Conversely, non-breeding Grey-faced Petrels arrive at the colonies typically 100–200 g lighter than birds starting an incubation shift (450–600 g versus 630–800 g for breeders) (Imber 1976, G. Taylor unpubl. data) and could thus afford to pursue less abundant or lower quality prey near the ocean surface compared with breeding birds. Another possibility is that Grey-faced Petrel breeders, though similar in body morphometrics to non-breeders (G. Taylor unpubl. data), have a much larger body mass due to rapid fat storage between incubation shifts. This may enable these birds to retain more body oxygen and insulate them against thermal loss which could enhance their diving performance.

## Effect of trip duration

The results of this study and others (Wanless *et al.* 1991) suggests that duration of MDG deployment can have a strong influence upon the results. Although maximum dive depths showed a positive increase with trip duration in this study, this was to be expected as the longer MDGs are deployed on birds, the more likely it is that deeper than average dives will be detected. In this study, an ANCOVA model was run to assess whether days to recovery of MDGs influenced the results of the sex and breeding status analysis. No covariate effect was detected.

Grey-faced Petrels proved to be a difficult species to sample diving behaviour. First, non-breeding birds only visit the colony for a few consecutive nights then spend long intervals back at sea (often greater than one week). The lengthy incubation shifts (13–21 days) (Imber 1976, Johnston & Davis 1990) and foraging spells (1–14 days) (G. Taylor unpubl. data) of breeding Grey-faced Petrels made it impractical to retrieve all MDGs after a single short foraging trip as recommended by Wanless *et al.* (1991) and Burger (2001). MDGs from breeding birds were mostly retrieved between 38 and 54 days after initial deployment (fig. 2, table 1). Males captured in burrows sitting on an egg in early July would only have spent one spell of 14–21 days at sea between early July and mid- to late August (Johnston & Davis 1990). The deepest maximum dive depths recorded in the Grey-faced Petrel trial came from MDGs retrieved after this lengthy deployment.

MDG recoveries from Flesh-footed and Sooty shearwaters were similarly affected by the length of incubation shifts and time needed to ensure an incubating bird had finished its shift, gone to sea and returned again to the colony. With the other species sampled in this study, the short duration of the field trips to the breeding colonies limited the opportunities to recover all of the MDGs deployed. MDGs were mostly recovered from birds 1–5 nights after deployment and birds away on longer trips were missed. If birds away on longer foraging trips dive deeper than those visiting the colony after several days, then the dive depths recorded in this study may underestimate the diving ability for some of these species.

## Implications of diving behaviour for fisheries interactions

Flesh-footed Shearwaters and Grey-faced Petrels are incidentally killed as bycatch by commercial fisheries in New Zealand, including on pelagic and demersal longlines (Robertson *et al.* 2004). The diving ability of these two common New Zealand breeding species has not been studied previously using MDGs. This study suggests that when

longlines are within 20–30 m of the surface, these seabird species can be at risk of capture. Longlines occur at such depths during setting and hauling, and are less frequently set to fish at such depths. Cherel *et al.* (1996) have shown that seabird species that are capable divers are able to recover the baited hooks on the sinking branch lines and return them to the surface where other larger and more aggressive species, e.g., Wandering Albatross, *Diomedea exulans* Linnaeus, 1758, have a chance to steal the bait and risk getting hooked. Setting longlines at night is now a preferred mitigation solution. This would give some protection to Flesh-footed Shearwaters, which forage by day (Marchant & Higgins 1990); however, such action may fail to protect the nocturnally active Grey-faced Petrel (Imber 1973) and other night-feeding petrels.

## How reliable are MDGs?

The basic assumption with using MDGs is that they provide an accurate assessment of the maximum depth reached by individual birds during foraging trips at sea. Burger & Wilson (1988) pointed out some potential sources of error when using capillary gauges including overestimating diving ability by behaviour such as plunge diving or repeated diving to maximum depths. They also considered that condensation or water retention inside tubes can also cause problems. The risk of overestimating dive depths by birds plunge diving was eliminated by pointing the open end of tubes towards the tail. Any faulty MDGs that retained water within the tubes were discarded in this study. Zavalaga & Jahncke (1997) tested the validity of results from MDGs by lowering nine MDGs from a research vessel into the sea between depths of 10–150 m. They found that MDG' overestimated real depths by an average of 13.2% (2.2–35%), with shallower depths (<70 m) being recorded more precisely (<8% error) than MDGs deployed to deeper depths (>70m). No explanation was offered by these authors as to why MDGs overestimated real depths. However, in the current study, only one dive was measured beyond 70 m so the MDG readings should be accurate within 10%.

Soil jammed into the open end of the tube was a new problem encountered in this study, particularly with data collected from breeding birds occupying clay soil burrows. As the burrow chambers tend to be small, the tail is often pressed into the sides of the nest chamber and soil was sometimes pushed into the first 5 mm of the open end of the tube. This issue is likely to be a significant variable when using MDGs as a method for studying diving behaviour in burrowing seabirds that nest in clay soils. If the dirt in the tube prevented seawater from entering and dissolving the sugar, some of the initial dives may possibly not have been registered by the MDG until the soil was dislodged. The presence of tiny grit particles within the tubes suggests this may have been a common occurrence with the breeding birds on Kauwahaia and Ihumoana islands, and is likely to result in an underestimate of diving performance if some deeper dives were missed. For example, a Grey-faced Petrel recovered 40 days after release still had soil blocking the end of the tube and the sugar was undissolved within the tube. For species that dive infrequently or only dive to shallow depths, water pressure may not be sufficient to dissolve or dislodge these dirt blockages. However, the dirt is more likely to be pushed up the tube, or gradually dissolved when exposed to greater water pressure, longer dive durations or increased frequency of immersion. These latter scenarios are more likely to occur with species that dive to deeper depths.

The sources of error noted above indicate that MDGs will provide only an estimation of diving ability in seabirds. Very small but expensive TDRs (3–6 g) are now becoming available for seabird research. For example, 6 g Lotek geolocation loggers with built-in pressure sensors were used on Sooty Shearwaters (n=11) and these confirmed that this species is able to reach a depth of at least 68 m (Shaffer *et al.* 2006), similar to the typical diving range recorded for this species in two studies using MDGs (see above). However, the lightweight capillary tubes used in this study still provide an inexpensive method to study the diving ability of many small and tiny seabird species weighing less than 500 g.

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(accepted 7 October 2008)

# WEDGE-TAILED SHEARWATERS, *PUFFINUS PACIFICUS*, AND DEVELOPMENT ON HERON ISLAND, GREAT BARRIER REEF, AUSTRALIA

by Johanna Rosier and Pam Dyer

(with three text-figures)

**Rosier, J. & Dyer, P.** 2008 (31:x): Wedge-tailed Shearwaters, *Puffinus pacificus*, and development on Heron Island, Great Barrier Reef, Australia. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 99–104. ISSN 0080-4703. Faculty of Arts and Social Sciences, University of the Sunshine Coast, Maroochydore DC, Queensland 4558, Australia. (JR*, PD). *Author for correspondence. Email: Jrosier@usc.edu.au

The Wedge-tailed Shearwater, *Puffinus pacificus*, colony on Heron Island, Great Barrier Reef, Australia, has been the subject of a long-term study since 1984 to monitor the effects of development. During this time, there has been considerable development in the tourist resort, university research station and national park ranger quarters. Several methods have been used to record and analyse the effects of development. Prior to the development of specialised burrow-viewing equipment, Wedge-tailed Shearwater population estimates had been based on indirect methods such as footprints and cleared burrow entrances. Investigation of actual burrow contents, relayed electronically from a camera inserted into burrows showed that all previous population estimates needed revision, even halving earlier populations estimates. We also used a photographic record and surveys to record the changes to island habitats over time. The Ultimate Environmental Threshold (UET) technique was used to estimate the effects of development on several environmental factors. Traditional breeding areas have been lost to the shearwater population through construction of buildings, paving of areas, blocking of flight paths and compaction of tracks. However, the species appears to have compensated despite significant modification to the island's natural habitats. Nesting behaviour has been altered in disturbed areas of the island and egg-hatching rates have been lowered. Because the life span of a shearwater could be up to 30 years, more long-term research is needed to analyse the true effects of development.

**Key Words: Heron Island, Great Barrier Reef, Wedge-tailed Shearwater, *Puffinus pacificus*, tourist resort, university research station, National Park, tourism, seabirds.**

## INTRODUCTION

The effects of tourism development on a seabird-breeding island on the Great Barrier Reef (GBR) in Australia may seem far removed from the interests of researchers concerned with the temporal and spatial effects of urbanisation on birds. However, we will demonstrate how many of the issues facing urban bird species (habitat fragmentation, edge effects, population stability, nesting success and the effects of human disturbance) also are present on Heron Island, despite its remoteness. Because the western half of the island is densely developed, the habitat of the Wedge-tailed Shearwater, *Puffinus pacificus* (J.F. Gmelin, 1789), can be likened to fringe areas of urbanised centres.

The Great Barrier Reef Marine Park Authority (GBRMPA) and the Queensland Parks and Wildlife Service (QPWS) are jointly responsible for managing the effects of human use of marine areas, especially the reefs. The island national parks are managed solely by the QPWS. The World Heritage-listed Great Barrier Reef Marine Park is managed on a multiple-use basis to achieve sustainable development. The GBR World Heritage Area (WHA) has a different boundary to the GBR Marine Park with different management requirements. For instance, all the islands are in the World Heritage area but not in the GBR Marine Park. They are part of Queensland's state lands and are part of Queensland. The main human activities are tourism, recreation, fishing and research. The islands provide bases from which most reef activities are carried out. Access points to the islands and the reef tourism sites are on the mainland (e.g., Cairns, Townsville, Mackay, Airlie Beach-Shute Harbour), with a few resorts mainly on continental islands.

Heron Island is one of the few resorts actually located on the Great Barrier Reef and is accessed from Gladstone. The island is a sand "cay" within the Capricorn/Mackay Section of the Great Barrier Reef Marine Park (GBRMP) (fig. 1) with a vegetated area of 13.5 ha (Hill & Barnes 1989, Dyer *et al.* 1995). QPWS measures note the area of the island as being 19 ha in total (Davie & Winter 1988) and that it is divided into three areas: the national park (QPWS – 12.3 ha), Heron Island Resort (HIR – Qld lease of 4.7 ha) and the Heron Island Research Station (HIRS – 2.3 ha), on land which has been leased from the QPWS by the University of Queensland for many years with the latest lease commencing in October 2002. An esplanade forms the remaining 2.3 ha of the national park. Approximately 75% of the land has national park status.

The resort lease is autonomous in that there are no conditions about development limitations on the lease documents. The lease agreement for the research station, which is part of the national park, stipulates limitations on permitted activities — so that they remain appropriate to the national park status of the area. When the research station's lease was renewed in 2002, additional conditions were also placed on its activities. Leaseholders have agreed to conform to the Heron Island Management Guidelines (Hunnam 1983, Davie & Winter 1988), which set limits on development, landscaping and activities by all parties on the island. Development of the island is also subject to the other guidelines (QPWS 2000).

The most striking feature of the cay's vegetation is the central core of *Pisonia grandis* R.Br. forest (Hill & Rosier 1989, Dyer & Hill 1992), which is surrounded by a fringe of wind/salt-tolerant vegetation consisting of *Argusia, Casuarina* and *Pandanus* spp, as well as stunted *P. grandis*. Another feature of the northern fringe area is the dense ground cover of grasses, many of which are introduced species.

The main ecological values of the marine park depend not only on the importance of the islands as breeding sites for turtles and a number of seabird species, but also on their

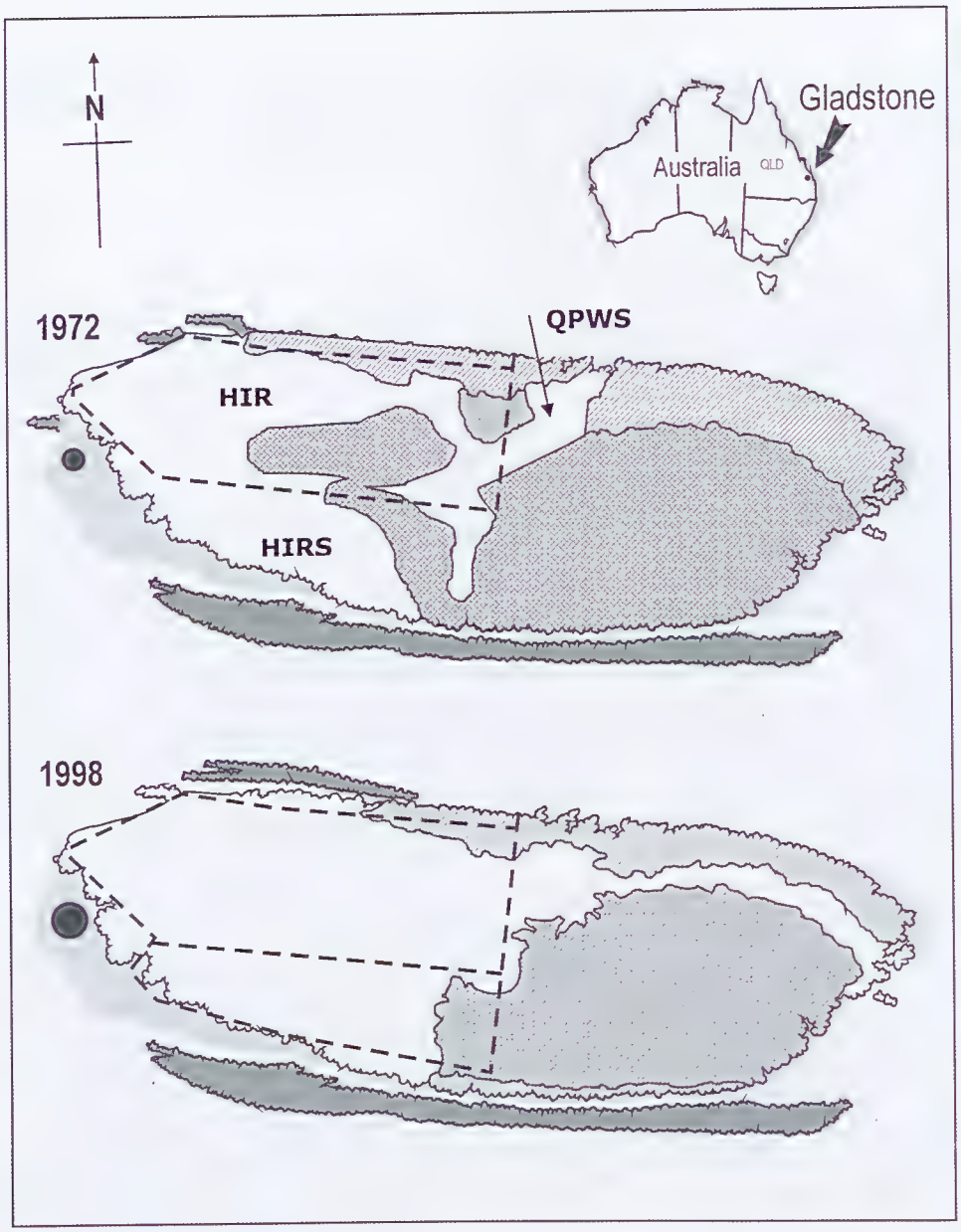


*FIG. 1 — Map of Heron Island changes 1972–1998.*

usefulness for educating people about the natural values of the island habitats and surrounding reef and tourism opportunities. In the 1800s and early 1900s, wildlife resources (e.g., nesting turtles, mainly *Chelonia mydas* (Linnaeus, 1758)) or wildlife-related resources (e.g., guano) have formed the basis of exploitative industries (QPWS 2000). All these types of activities involve manipulation of the resource base, albeit with differing degrees of impact. However, even relatively benign development, such as the establishment of a scientific research station or ranger station, requires that decisions about the location of facilities and activities be based on a spatial analysis of environmental variables to achieve the sustainable use of resources and avoid potential, irreversible impacts on the environment.

The human population of the island is difficult to assess, particularly in the resort area. HIR regards this information as commercially sensitive. Ferrier (1984) noted that the island population varied between 400 and 600 people (based on an annual turnover rate for visitors in the vicinity of 10 000 guests per year). This figure included tourist resort guests and staff, HIRS research and maintenance personnel and QPWS officers stationed in the National Park Headquarters. By 1986 the resort recorded 49 000 visitors yearly, and by March 1987, the resort was already catering for 1200 guests per week or 60 000 visitors on an annual basis. The research station accommodates up to 84 researchers and students, and the QPWS base houses up to 11 people mainly during turtle nesting season. Increasing demands for development space in each leased area is impacting on the available space for Wedge-tailed Shearwater breeding. The island management committee and QPWS have no formal framework for monitoring the impacts of development on any part of the environment. Current monitoring of development is focused on managing the national park area of the island rather than the leased areas. Following the fire in 2007, the research station is establishing a more comprehensive database to record all research sites on the island.

## METHOD

In view of the fact that shearwater nesting areas (and those of other species) need to be protected (Kozlowski 1985), the Ultimate Environmental Threshold (UET) method

was applied to Heron Island to determine areas from which development and activities should be excluded, if development was happening for the first time (Rosier *et al.* 1986, Kozlowski *et al.* 1988, Hill & Rosier 1989). UET evaluates the quality of environmental elements relative to the consequences of development and use. To define UET thresholds to development, each of the relevant elements on Heron Island (geomorphological structure, vegetation, seabirds and turtles) was assessed according to its ecological significance or rarity, resistance or ability to cope with human disturbance, and transformation or degree of change "by humans" from its pristine state. The information gained facilitated assessment of the spatial compatibility between current development on Heron Island (across all lease areas and the national park) and the Wedge-tailed Shearwater breeding areas. The results were also used to retrospectively critique decisions about development. A series of planning inspections and photographs taken over 14 years provide a quantifiable record of development over time.

We also monitored breeding populations of Wedge-tailed Shearwaters which are monogamous pelagic seabirds. On Heron Island, they nest in burrows located in areas likely to be impacted by development (i.e., in vegetated areas above the Mean High Tide mark beyond turtle nesting areas). We visited Heron Island in both August and November/December each year from 1984–1987, in December 1988–1991, and in January 1998 and 1999.

The Heron Island Wedge-tailed Shearwater nesting population has been surveyed regularly since 1983. Prior to the development of specialised burrow-viewing equipment (Dyer & Hill 1991, Dyer & Aldworth 1998), population estimates were based on indirect methods such as footprints, cleared burrow entrances, and fresh guano. This was inefficient as non-breeding birds utilise empty burrows leaving the impression that the burrow supports breeding activities. Conversely burrows that have entrances covered by leaves, seemingly devoid of nesting activity, are just as likely to be occupied by breeding birds (Dyer 1990, 1991). Thus, population estimates based on burrow counts needed to be adjusted according to breeding activities occurring within the burrows.

Shearwater population censuses (1985–1990 and 1999) were derived by collecting data concerning the number of burrows, burrow status and different habitat types (e.g., Disturbed, Fringe areas, *P. grandis* forest). Figure 1 shows the habitat changes since mapping began in 1972. Data were collected for contiguous quadrats, forming systematically spaced transects that traversed the island in a north–south direction (Dyer & Hill 1990, Dyer 1992, Hill *et al.* 1995, 1996, Dyer *et al.* 2005). Following the introduction of the burrowscope, Carter *et al.* (1996a, b) and Carter (1997) investigated rates of breeding success at the hatching and fledgling stages for shearwater chicks for the 1993/4 breeding season. The study focused on analysing the environmental variables such as soil moisture, nest density and soil strength. Selected burrows were monitored four times during the nesting season, with timing coinciding with egg-laying, hatching, fledging and departure of fledglings. Linear Structural Relations (LISREL) models were applied to one-off habitat measurements to determine the most likely influences of various variables on breeding success.

In addition, Dyer *et al.* (1995, 2005) attempted to rationalise all previously published population estimates for Wedge-tailed Shearwaters in the Capricorn Group. Most population estimates had been established by doubling burrow counts, assuming a breeding pair for each burrow (fig. 2). Where available, data other than those provided by the authors were re-worked to standardise earlier population estimates with those of Dyer, Hill, Rosier and Carter (fig. 2). The outcome, though relatively crude, was considered an improvement over data taken at face value for comparative purposes. However, because incubation rates of about 50% were not significantly different between two seasons, it was proposed that all previous population estimates based on burrow counts should be at least halved (Dyer 1992).

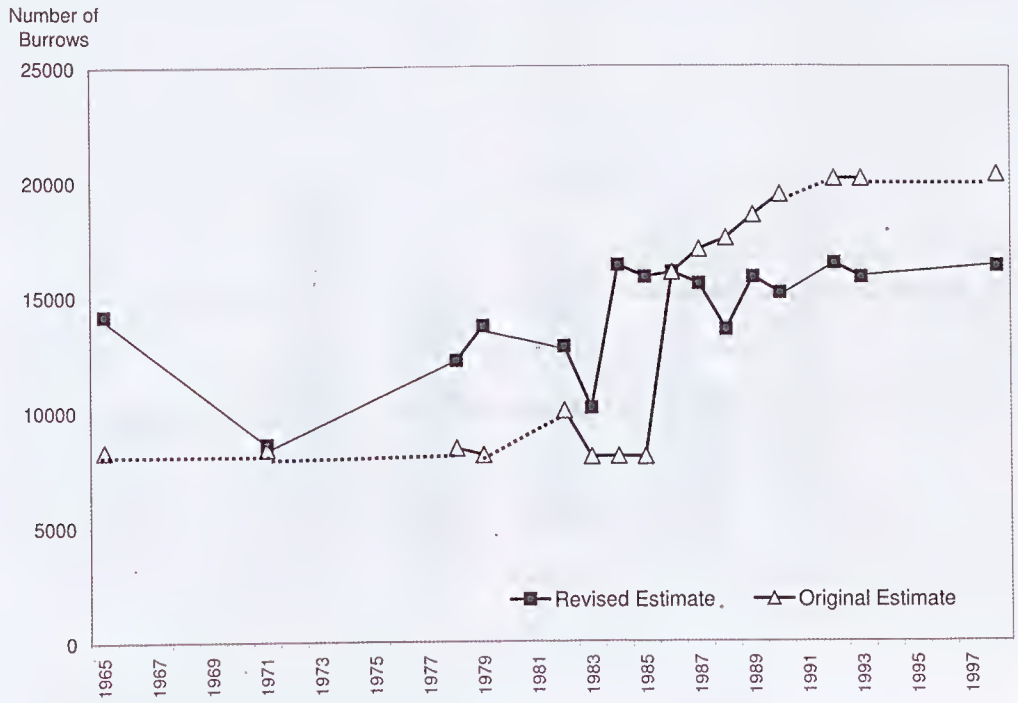


*FIG. 2 — Rationalisation of Wedge-tailed Shearwater burrow estimates.*

## RESULTS

The UET analysis determined that before 1988, only the research station buildings had a significant effect on potential nesting areas for shearwaters. The resort had already been constructed and the habitat area virtually lost (Hill & Rosier 1989). In addition, the habitat available to birds on the northern side of the island is less favourable because of the dry, sandy soil. Neil & Dyer (1992:37) found that "burrowing density tends to increase into the mid part of the soil strength range ... favouring sites where the soil is loose enough to burrow easily, but not so loose as too collapse" (the dry sandy soils in the resort area). Subsequent development (sewerage treatment station and new road between the resort and the QNPWS station) has compromised ecologically sensitive areas in the centre of the island, characterised by closed *P. grandis* forest. By 1998, only 30% of the *Pisonia* forest remained and only 11% of the fringe area remained relatively undisturbed. Figure 3 shows the changes which have occurred in habitat areas over the years that management plans were prepared. The UET analysis assumed that prior to permanent human occupation in the 1960s, *P. grandis* forest was the preferred habitat type for shearwater breeding areas. However, subsequent monitoring on Heron Island demonstrates that shearwaters nest in both the *Pisonia* forest and fringe areas bordering the forest (Rosier *et al.* 1986, Kozlowski *et al.* 1988).

Based on the number of shearwater burrows on Heron Island, the Wedge-tailed Shearwater population had fluctuated around a mean of about 15 000 burrows for the 35 years to 1995. The results of the annual surveys (1985–1989) showed no significant trends in the burrow estimates for the entire island or for either the western (disturbed) or eastern (least disturbed) sides of the island. By December 1989, there was a significant difference in overall burrow numbers between the eastern and western sides [$t_{18} = 2.1$, $P < 0.05$ (independent two-tailed test)] of the island, but this was no longer evident in December 1990, suggesting habituation by shearwaters (Hill *et al.* 1995). A preliminary 1999 census resulting in about 16 500 burrows overall is within normal parameters but significantly fewer burrows were found between non-developed and developed areas (5:3), that is between eastern and western sides of the island [$t_{12} = 2.2$, $P < 0.05$ (independent two-tailed test)]. It appears that birds have now compensated for the loss of habitat in developed areas by nesting at higher densities in the national park. Only when current research is complete will it be clear whether this pattern of shearwater nesting has now become more permanent or whether this is a temporary aberration similar to the findings for December 1989.

Burrow occupancy rates between December 1989 (43%) and December 1990 (49%) were not significantly different between seasons for the island as a whole ($\chi^2 = 1.75$, df = 1, $P > 0.1$), or for the individual habitats types ($\chi^2 = 2.6$, df = 4, $P > 0.6$). Although the buildings category of nesting area supported occupancy rates of 52% in December 1989 and 44% in December 1990 ($\chi^2 = 0.86$, df = 1, $P > 0.3$) (Dyer 1992, Hill *et al.* 1995), the difference was not significant. The developed habitat had low burrow densities ranging from 0.015 to 0.031 per $m^2$ between 1985 and 1990. The *P. grandis* forest and fringe habitats, together covering about 50% of the island for the 1980s studies provide the main nesting habitats for Wedge-tailed Shearwaters. These support between 60% and 78% of the burrows depending on the year (Hill *et al.* 1995). This habitat has been reduced so much that it only covered approximately 41% of the island in January 1998 (figs 1, 3) and supported about 84% of total burrows.

Hill *et al.* (1996) show that the disturbed habitat on Heron Island has significantly lower hatching success (56.76%) compared with up to 80% hatching success in the fringe and *Pisonia* forest ($\chi^2 = 13.186$, df = 3, $P < 0.005$). Substrate soil moisture and nesting density were found to be the major factors affecting breeding success of shearwaters on Heron Island (Carter 1997). For example, low substrate soil moisture is needed while nests are being excavated, but higher soil moisture is needed to ensure successful hatching of fledglings. However, further research is needed before we can show whether it is moisture elements that cause low breeding success in the disturbed area.

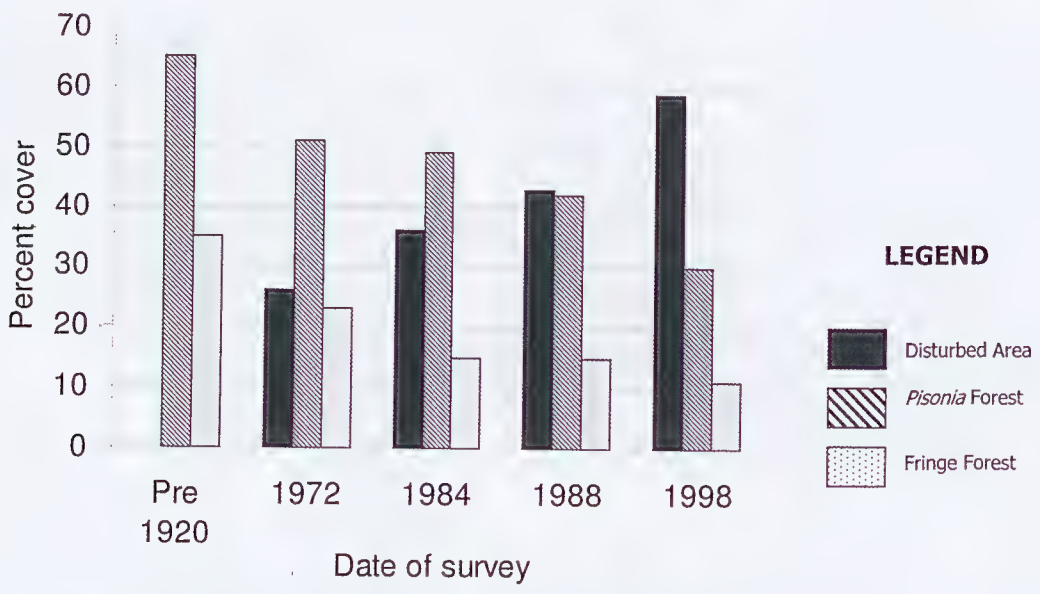


*FIG.3 — Changes to Heron Island habitat type with increased development.*

## DISCUSSION

### Management implications

Heron Island has undergone substantial change, especially since the 1960s when the resort was established. Most change has been unplanned in terms of anticipating and understanding the effects of development on shearwater nesting areas. The cumulative spatial impacts of small incremental building projects would not have been tolerated had the total development been proposed as a single development project. None of the planning mechanisms to limit development or activities (esplanade and buffer zoning, total human population limits, height and extent of building limits, restriction of vehicular access or prohibition of activities) has been implemented consistently over time. The results of this research highlight the effects of increased development on shearwater nesting density, breeding success, inter-island movement of seabirds and vulnerability of ground-nesting seabirds.

Shearwater nesting densities on Heron Island are consistently lower compared with corresponding areas on North West and Masthead islands (Hulsman 1983). The problem is determining the degree to which human disturbance has caused lower nesting densities and the subsequent effect of those lower nesting densities on breeding success. Construction of structures, dumping of building products in the national park area of *Pisonia* forest and the increased compaction of areas such as roads, paths and recreation areas preclude nesting by shearwaters. Buildings now block half of the shearwater flight paths mapped in 1983. There also has been little research into the patterns of nesting, an aspect that appears to be important to birds nesting synchronously (Nelson 1980).

Breeding success rates of Wedge-tailed Shearwaters on Heron Island are still not known (Carter *et al.* 1996b), although shearwaters seem to be relatively resilient. They appear to have responded to exclusion from earlier sites by burrowing in higher densities or moving to other parts of the island. It cannot, however, be assumed that altered nesting patterns are not in themselves detrimental to breeding success rates. Therefore, after 14 years of research it is still difficult to determine conclusively that development will impact negatively on the shearwater population over the long term. Research about breeding success rates over time are needed so that the real effect of human impacts can be determined. The opportunity to monitor occupancy rates has not arisen since Carter's 1993/4 study.

The Capricorn Group of islands has several shearwater breeding islands in addition to Heron Island. Management plans should include analysis of the relationship between colonies and indicators for monitoring human impacts on seabird-nesting areas across the island group. Ideally, protection of seabird colonies requires critical areas be set aside as sanctuaries; access is controlled; information about the birds and restrictions on access is readily available; and the effects of human disturbance are understood (Segnestam 1975, Anderson & Keith 1980, Feare 1984, WBM Oceanics & Claridge 1997). To achieve this, the following questions need to be answered.

What is the acceptable area and level of development in the long term?

What constraints must be imposed on development and people's activities to ensure that natural areas are managed effectively to ensure that reproductive success of seabirds is not threatened?

One of the major research problems in the Capricorn Group of islands is the lack of knowledge about seabird movement between islands. Therefore, any policy that relies on sacrificing one island for development (QPWS 1980) may have detrimental long-term effects on the island group. When researching the characteristics of the colonies, few scientists have commented on the status of other colonies throughout the world. This also may affect decisions concerning development on islands of seemingly minor importance in the region. The data sets required to provide quantitative assessments of the significance of the colonies and the effects of human disturbance are extensive and involve in-depth understanding of total ecosystems (e.g., worldwide distribution of a species) as well as their component parts (e.g., local distribution and ecology of a species). As Hulsman (1983) points out, it may take up to ten years to gather relevant data and this is conditional on a decision to do so by management authorities.

The effects of human activities on ground-nesting seabirds are particularly difficult to evaluate (Anderson & Keith 1980). Resistance of Wedge-tailed Shearwaters may be gauged superficially only due to a lack of research into this aspect of their ecology. Nelson (1980) reports that disturbed sites are less attractive as nesting areas although this does not appear to be a universal phenomenon (Hill & Barnes 1989, Dyer & Hill 1990, Carter 1997). Since Shearwaters lay only one egg per season, any damage to the nesting burrow during the nesting season will affect success. The physical layout of burrows is such that severe damage occurs if humans enter an area with high burrow densities. This applies to both breeding and non-breeding seasons (Nelson 1980).

There are two major conclusions from our ongoing study. First, there has been a major reduction of shearwater breeding habitats on Heron Island since 1972 when mapping of habitat and development first occurred. Development decisions have been made on an ad-hoc basis with no overall strategy guiding tourist or research station development. All plans and monitoring of development have not taken into account the long-term protection of ecological values and the natural characteristics that attract and educate tourists, especially in relation to the ground-nesting shearwaters.

Second, current results demonstrate that, despite the foreboding/concerns of various researchers (Hulsman 1983, Hill & Rosier 1989), the effects of development on the shearwater population have to date been minor. Alternatively, it may be that potential adverse effects implied by the lower hatching rates in the developed areas are not yet apparent. Despite the significant modification of Heron Island's natural environment and despite human activities on nearby uninhabited islands, Wedge-tailed Shearwaters in the Capricorn Group of islands appear to have acclimated to the presence of humans. The degree of impact over the long term will be difficult to gauge without understanding the optimum levels of breeding populations, research that requires many years of work. Shearwaters do not return to breeding colonies for five years after fledging and the life span of shearwaters could be up to 30 years. All these factors need to be factored in to modelling of human impacts.

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(accepted 5 August 2008)

# NOTES ON RARE, VAGRANT AND EXOTIC AVIFAUNA AT MACQUARIE ISLAND, 1901–2000

by G. R. Copson and N. P. Brothers

(with one text-figure and three tables)

**Copson, G. R. & Brothers, N. P.** 2008 (31:x): Notes on rare, vagrant and exotic avifauna at Macquarie Island, 1901–2000. *Papers and Proceedings of the Royal Society of Tasmania* **142**(1): 105–116. ISSN 0080-4703. 32 Red Chapel Ave, Sandy Bay, Tasmania 7005, Australia (GRC*), 176 South Arm Drive, Wonga Beach, Queensland 4873, Australia (NPB). *Author for correspondence.

The occurrence and status of rare, vagrant and exotic bird taxa recorded at Macquarie Island between 1901 and 2000 are reviewed. The number of species recorded at the island, excluding those that have bred regularly on the main island between 1901 and 2000, is increased to 56 non-breeding species, seven species that have been confirmed breeding since 1980 and four introduced/exotic species breeding on the island.

**Key Words: Macquarie Island, sub-Antarctic, avifauna records.**

## INTRODUCTION

Sub-Antarctic Macquarie Island (fig. 1) lies in the Southern Ocean, at 54°30'S, 158°57'E, approximately 1500 km SSE of Hobart, Tasmania, and 1100 km SSW of Invercargill, New Zealand. The nearest land is the Auckland Islands, 600 km NE and Campbell Island 650 km ENE. The island is 34 km long by 5.5 km at its broadest point and 12 800 ha in area. There are two small outlying groups, the Judge and Clerk islets 11 km to the north and the larger Bishop and Clerk islets 37 km to the south, together with several offshore sea stacks around the main island. Macquarie Island rises steeply from the sea to an undulating plateau, ranging between 200 m and 300 m above sea level, having a maximum altitude of 433 m. The northwestern part of the island is fringed by a raised beach-terrace up to 800 m broad and 15 m above sea level.

The climate is extreme oceanic with a mean monthly temperature of 3.8°C, precipitation on 312 days of the year with an annual mean of 920 mm, average humidity of 89% and an average wind speed of 33.4 kph with over 70% of the wind coming from within the southwest to northwest quadrant (De Lisle 1965, Selkirk *et al.* 1990).

Since its discovery by sealers in 1810, several species of exotic fauna have been either deliberately or accidentally introduced onto the island. By the end of the nineteenth century the Feral Cat, *Felis catus* Linnaeus, 1758, Weka, *Gallirallus australis* (Sparrman, 1786), European Rabbit, *Oryctolagus cuniculus* (Linnaeus, 1758), House Mouse, *Mus musculus* Linnaeus, 1758, and Black Rat, *Rattus rattus* (Linnaeus, 1758), had become established (Cumpston 1968, Jenkin *et al.* 1981). In addition, other exotic species became established on the island after dispersing from their introduction sites in Australia and/or New Zealand. In the twentieth century Common Redpoll, *Acanthis flammea* (Linnaeus, 1758), Common Starling, *Sturnus vulgaris* Linnaeus, 1758, and Northern Mallard, *Anas platyrhynchos platyrhynchos* Linnaeus, 1758, became established on Macquarie Island in such a manner (Falla 1937, Norman 1987).

The direct impact of the sealers combined with the introduction of terrestrial vertebrates led to major changes in the status of species of the indigenous flora and fauna. The earliest detailed studies of the fauna occurred with Scott's

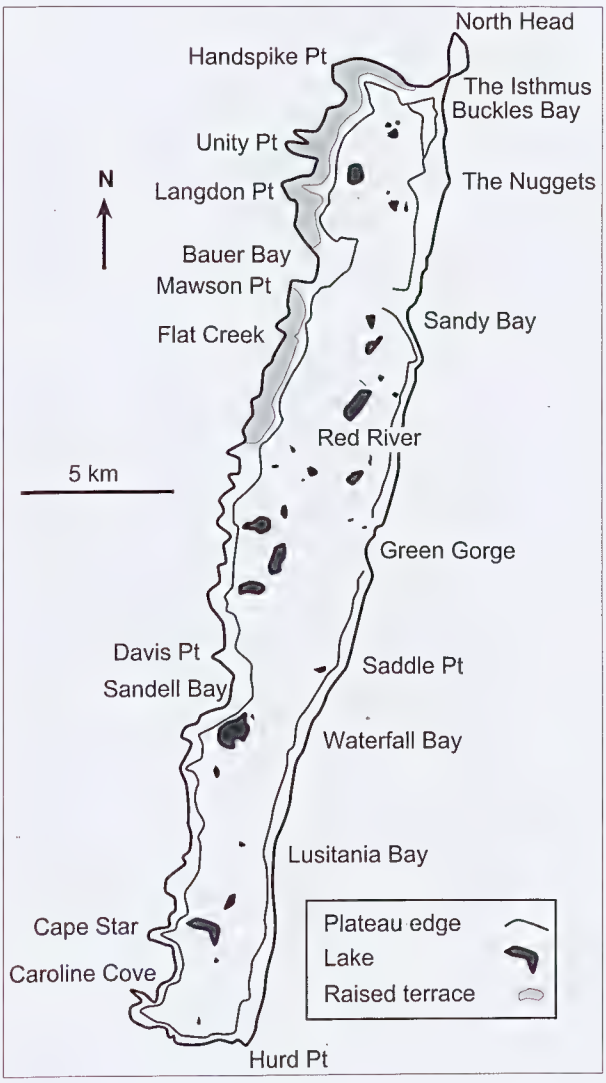


*FIG. 1 — Macquarie Island localities referred to in the text.*

visit in 1880, Burton's records of the avifauna between 1896 and 1900 and H. Hamilton's work during the Australasian Antarctic Expedition 1911–14 (AAE), i.e., after several of the exotic species had become well established (Scott 1883, Campbell 1901, Mawson 1930, Falla 1937). Little is known therefore of the island's flora, fauna and ecosystems in their pristine state.

The establishment of an Australian National Antarctic Research Expeditions (ANARE) Station on the island in 1948 has resulted in studies, of the highest order, in a broad range of scientific disciplines including the biological sciences. In addition the continuous staffing of the island since 1948 has resulted in the accumulation of incidental observations which will, given a sufficient time span, become valuable data. The bird records below are an example of this.

## METHODS

The records are all from the period 1901–2000 and include a wide range of published and unpublished sources. It was considered that 2000 was an appropriate time to finish the review as the vertebrate pest management program being undertaken, together with climate change, are likely to have increasing impacts on the status of both the indigenous and vagrant avian species at the islands over the next decades. During the period covered Macquarie Island was occupied by sealers/penguin oil industry workers intermittently up to 1920, the AAE and Australian Bureau of Meteorology staff from 1911–15 and ANARE personnel continuously from 1948. The great majority of the records are from the AAE, effectively 1912–13, and ANARE periods covering approximately 54 years.

Nineteen avian species that are considered native to Macquarie Island and known to have bred more or less continuously on the island over the period considered are not included in the review. Species whose breeding has been proven only in the last few years of the records are included. All exotic species, i.e., the deliberately introduced Weka together with the self-introduced northern hemisphere species, have been included whether breeding or not on the island. Domestic stock taken to the island during the AAE and some early ANARE years are not included.

In order to give an indication of the frequency of visitation for each taxon, tables 1–3 show the number of years in which they have been recorded at the island.

Numbers in brackets following each note indicate the source of the record. These are: 1. Hamilton 1912–13; 2. Falla 1937; 3. Serventy 1952; 4. Lindholm 1952; 5. Gwynn 1953; 6. Keith & Hines 1958; 7. Gibson 1959; 8. Warham 1961; 9. Simpson 1965, 10. Warham 1969; 11. Merilees 1971a; 12. Merilees 1971b; 13. Bourne 1981; 14. Brothers 1984; 15. Brothers & Ledingham 2008; 16. Brothers & Bone 2008; 17. Unpublished ANARE, Tasmanian Parks and Wildlife Service (PWS) and Department of Primary Industries and Water logbooks and records.

In the text any reference to Macquarie Island includes the off-lying sea stacks, the Judge and Clerk islets and the Bishop and Clerk islets unless otherwise indicated. Observations from re-supply vessels out to 10 km have been included.

The systematics and taxonomy largely follows Christidis & Boles (2008). All records have been entered onto a spreadsheet including, as far as possible, the date, location, observer/s, numbers recorded and any notes for each taxon. Copies of the spreadsheet will be lodged with the Australian Antarctic Division, Tasmanian Department of Primary Industries and Water and Tasmanian Parks and Wildlife Service.

## RECORDS 1901–2000

### Vagrant and regular visiting species

#### Australian Shelduck, *Tadorna tadornoides* (Jardine & Selby, 1828)

A single female Australian Shelduck was sighted amongst tussock in the Caroline Cove area in December 1990. In 1998, on 30 January, a male and three females were sighted on a small lake in Red River and on 13 February a male and four females were seen at the same site [17].

#### Grey Teal, *Anas gracilis* Buller, 1869

While the first positive record was in 1957 there appear to have been several other unrecorded sightings of this species between 1948 and 1957. There are also three records from 1979 when three birds were seen in April and one in May at Green Gorge and one sighted in the Red River area in September [6, 17].

#### Hoary-headed Grebe, *Poliocephalus poliocephalus* (Jardine & Selby, 1927)

In 1975 sightings of single birds were made in August (2), September (1) and October (2) while in November two birds were seen together. All sightings were at the same site, a small pool at Green Gorge [17].

#### Rock Dove, *Columba livia* J.F. Gmelin, 1789

This is a record of a banded bird which was released in a race from Victoria to Tasmania on the morning of 29 October 1989 and caught at the ANARE Station, on The Isthmus, on 31 October 1989 [17].

#### Fork-tailed Swift, *Apus pacificus* (Latham, 1802)

On 11 December 1958 a Swift was seen over the Royal Penguin colonies at The Nuggets but could not be positively identified. On 12 December a clear view of a Fork-tailed Swift was obtained around North Head. The only other records of this species were of one seen on 27 November 1975 at Perseverance Bluff, the southern end of The Isthmus, and one seen about 2 km north, in the Secluded Beach area, on the next day [7, 17].

#### White-throated Needletail, *Hirundapus caudacutus* (Latham, 1802)

Between 1960 and 1992 there were 25 sightings, totalling over 46 birds, of this species at Macquarie Island. The first record is of one collected in January 1960. The next records were in the summer of 1969–70 when seven sightings each of a single bird were recorded between 26 December 1969 and 10 February 1970. From 19 January to 11 February 1971 there were ten sightings of up to six birds recorded. Further sightings occurred in 1977 (1), 1979 (2), 1980 (1), 1985 (1) and 1992 (3). While the majority of these were in January or February one bird was sighted just south of The Nuggets on 9 March 1977 and two at Hurd Point on 24 April 1985 [8, 10, 17].

**TABLE 1**
**Non-breeding birds recorded at Macquarie Island, 1901–2000**

| Species | Number of birds recorded in month[1] | | | | | | | | | | | | No. years recorded |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | J | F | M | A | M | J | J | A | S | O | N | D | |
| Australian Shelduck | 4 | 5 | – | – | – | – | – | – | – | – | – | 1 | 2 |
| Grey Teal | – | – | – | 3 | 1 | – | – | – | 1 | – | – | 1 | 2 |
| Hoary-headed Grebe | – | – | – | – | – | – | – | 2 | 1 | 2 | 2 | – | 1 |
| Emperor Penguin | – | 1 | 1 | – | – | – | – | – | – | – | – | – | 2 |
| Adelie Penguin | 1 | – | – | – | – | – | – | – | – | – | 1 | 1 | 3 |
| Chinstrap Penguin | 8 | 4 | 5 | – | – | – | – | – | – | 1 | 14 | 7 | 23 |
| Snares Penguin | 2 | 3 | 1 | – | – | – | – | – | – | – | 1 | – | 4 |
| Erect-crested Penguin | 8 | 25 | 8 | 2 | 1 | – | – | – | – | 6 | 2 | 4 | 23 |
| Southern Fulmar | 2 | – | – | – | – | – | – | – | 6 | 5 | 7+ | 6 | 12 |
| Antarctic Petrel | 5 | – | – | – | – | – | – | 1 | 1 | 1 | 4 | 1 | 5 |
| Snow Petrel | – | – | – | – | – | 1 | 1 | 2+ | – | – | – | – | 3 |
| Soft-plumaged Petrel | 2 | – | – | – | – | – | – | 1 | – | 1 | 16+ | 2 | 4 |
| Mottled Petrel | 1 | – | – | – | – | – | – | – | – | – | – | – | 3 |
| Slender-billed Prion | – | – | 1 | – | 1 | 1 | – | – | 1 | – | 1 | – | 3 |
| Westland Petrel | 1 | – | – | – | – | – | – | – | – | – | – | – | 1 |
| Short-tailed Shearwater | 2 | – | – | 1 | – | – | – | – | – | – | 4 | – | 4 |
| Shy Albatross | 1 | – | – | – | – | – | – | – | – | – | – | 1 | 1 |
| Salvin's Albatross | – | – | – | – | – | – | – | – | – | – | – | 1 | 1 |
| Sooty Albatross | – | – | – | – | – | – | – | – | – | – | 1 | 1 | 2 |
| Grey-backed Storm Petrel | – | – | – | 1 | – | – | – | – | 4 | – | 2 | – | 4 |
| Australian Gannet | 1 | – | – | – | – | – | – | – | 1 | – | – | – | 2 |
| Great Cormorant | 68 | 48+ | 66 | 4 | 1 | – | – | – | – | 12 | 47 | 94+ | 7 |
| White-faced Heron | – | 1 | 8 | 7+ | 17 | 7 | 2 | 5 | 4+ | 1 | 1 | 3 | 10 |
| Little Egret | – | – | – | 2 | – | – | – | – | – | – | – | – | 2 |
| Eastern Great Egret | – | – | – | 5 | – | 1 | – | – | – | – | – | – | 3 |
| Intermediate Egret | – | – | 2 | – | 1 | – | – | – | – | – | – | – | 1 |
| Cattle Egret | – | – | – | 15 | 1 | 1 | – | 1 | – | – | – | – | 9 |
| Swamp Harrier | – | 1 | – | – | – | 1 | – | – | 1 | 2 | 1 | – | 4 |
| Baillon's Crake | 1 | – | – | – | – | – | – | – | – | – | – | – | 1 |
| Eurasian Coot | 1 | – | – | – | 1 | 2 | – | 2 | 2 | 2 | – | – | 5 |
| Latham's Snipe | 1 | – | – | – | – | – | – | – | – | – | 2 | 3 | 6 |
| Bar-tailed Godwit | – | – | – | – | – | – | – | – | – | 1 | – | 1 | 2 |
| Common Greenshank | – | – | – | – | – | – | – | – | – | – | – | – | 1 |
| Ruddy Turnstone | 4 | – | – | – | – | – | – | – | ? | 1 | – | 25 | 5 |
| Great Knot | – | – | – | – | – | – | – | – | – | 1 | – | – | 1 |
| Red Knot | – | – | – | – | – | – | – | – | – | – | 1 | 6 | 2 |
| Red-knecked Stint | – | – | – | – | – | – | – | – | – | 4 | – | – | 1 |
| Curlew Sandpiper | 2 | – | – | – | – | – | – | – | – | – | – | – | 1 |
| Red-knecked Phalarope | 2 | – | – | – | – | – | – | – | – | – | – | 3 | 2 |
| Pied Oystercatcher | 1 | – | – | – | – | – | – | – | – | – | 3 | – | 2 |
| Black-winged Stilt | – | – | – | 1 | 14 | – | – | – | – | – | – | – | 1 |
| Grey Plover | – | 1 | – | – | – | – | – | – | – | – | – | 1 | 2 |
| Lesser Sand Plover | 7 | – | – | – | – | – | – | – | – | – | – | – | 1 |
| Black-fronted Dotterel | – | – | – | – | – | – | – | – | 2 | 1 | 1 | – | 3 |
| Masked Lapwing | – | 2 | – | – | 4 | – | – | – | 18 | 3 | ? | 5 | 6 |
| Arctic Tern | 3 | – | – | – | – | – | – | – | – | – | 4 | – | 3 |
| Rock Dove | – | – | – | – | – | – | – | – | – | 1 | – | – | 1 |
| Pallid Cuckoo | – | – | – | – | – | – | – | – | 1 | – | – | – | 1 |
| White-throated Needletail | 30+ | 11+ | 1 | 2 | – | – | – | – | – | – | – | 2 | 9 |
| Fork-tailed Swift | – | – | – | – | – | – | – | – | – | – | 2 | 1 | 2 |
| European Goldfinch | – | – | – | 1 | 3 | – | – | – | – | – | – | – | 3 |
| Welcome Swallow | 1+ | – | ? | 25 | 4 | – | – | – | 2 | 3 | 5 | – | 9 |
| Tree Martin | – | – | – | – | – | – | – | – | 1 | 1 | – | – | 2 |
| Silvereye | – | – | – | – | 2 | – | – | – | 1 | 1 | 1 | – | 2 |
| Common Blackbird | – | – | 1 | 4 | 4 | 3 | 4+ | 11 | 8 | 1 | 3 | – | 11 |
| Song Thrush | – | – | – | – | 1 | – | 4 | 8 | 7 | 2+ | – | – | 8 |

[1] Numbers of birds recorded each month include re-sights.

**TABLE 2**
**Rare and exotic breeding birds at Macquarie Island, 1901–2000**

| Species | Number of birds recorded in month[1] | | | | | | | | | | | | No years recorded | Status[2] |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | J | F | M | A | M | J | J | A | S | O | N | D | | |
| Northern Mallard | Recorded throughout the year | | | | | | | | | | | | 31 | b |
| Common Diving-Petrel | 1 | – | 3 | 5 | 2+ | – | – | – | 2 | 1+ | – | – | 13 | b |
| South Georgian Diving-Petrel | – | – | – | – | 1 | – | – | – | 2 | – | – | 4 | 3 | b |
| Cape Petrel | 2 | 5 | 2 | 8+ | 70 | 6 | 11 | 117+ | 145+ | 147+ | 91+ | 14+ | 23 | b |
| Fairy Prion | – | – | – | – | ? | – | – | – | 2+ | 2+ | – | – | 3 | b |
| Fulmar Prion | – | – | 1 | – | – | – | – | 5 | – | – | – | – | 2 | b |
| Grey Petrel | – | – | 3 | 8 | 5 | 2 | – | 1 | – | – | 1 | – | 10 | b |
| Wilson's Storm-Petrel | – | – | – | – | – | – | – | – | – | – | – | 4 | 1 | b |
| Weka | Recorded throughout the year until 1988 | | | | | | | | | | | | 43 | ext |
| Common Redpoll | Recorded throughout the year | | | | | | | | | | | | 54 | B |
| Common Starling | Recorded throughout the year | | | | | | | | | | | | 52 | B |

[1] Numbers of birds recorded each month include re-sights.
[2] B = more than 150 breeding pairs; b = fewer than 150 breeding pairs; ext = extinct.

**TABLE 3**
**Doubtful bird identifications at Macquarie Island, 1901–2000**

| Species | Number of birds recorded in month | | | | | | | | | | | | No years recorded |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | J | F | M | A | M | J | J | A | S | O | N | D | |
| Double-banded Dotterel | – | – | – | – | – | – | – | – | – | 1 | – | – | 1 |
| Hooded Plover | 4 | – | – | – | – | – | – | – | – | – | – | – | 1 |
| Little Tern | – | – | – | – | – | – | – | – | – | – | – | 1 | 1 |
| Richard's Pipit | – | – | – | – | – | – | 1 | – | – | – | – | – | 1 |

### Grey-backed Storm-Petrel, *Garrodia nereis* (Gould, 1841)

Grey-backed Storm Petrels have been recorded at Macquarie Island five times, seven birds in all. A female in breeding condition was collected in September 1956 while in 1979 and 1990 birds were attracted to the lights on The Isthmus and released after identification. Two were recorded in 2000 [6, 14, 15].

### Shy Albatross, *Thalassarche cauta* (Gould, 1841)

The first sighting, January 1989, was from the stern of the re-supply vessel, MV *Polar Queen*, approximately 5 nm northwest of the ANARE Station. The ship was manoeuvering slowly, while searching for a wave recorder buoy and had attracted several species of birds close up. In December 1989 the species was again recorded scavenging around Killer Whales, *Orcinus orca* (Linnaeus, 1758), close inshore at Sandy Bay [17].

### Salvin's Albatross, *Thalassarche cauta salvini* (Rothschild, 1898)

The Bishop and Clerk islets, 37 km south of the main island, were visited for only the third time in December 1993. During this visit a Salvin's Albatross was seen amongst the Black-browed Albatross, *Thalassarche melanophris* (Temminck, 1828), which breed on the islets [15].

### Sooty Albatross, *Phoebetria fusca* (Hilsenberg, 1822)

Single birds were recorded in December 1975 and November 1978. The 1975 bird had landed on the southern slopes of Cape Star, on the west coast, while the 1978 record was of a bird 0.5 km offshore around Douglas Point [17].

### Southern Fulmar, *Fulmarus glacialoides* (A. Smith, 1840)

Southern Fulmars were recorded at Macquarie Island in 12 separate years between 1956 and 2000. Sightings and collections were from many sites around the island. Ten of the 24 records were of beach-washed material and specimens, with most others being of birds flying close inshore. In December 2000 a bird was caught at The Isthmus and released later [6, 17].

### Antarctic Petrel, *Thalassoica antarctica* (J.F. Gmelin, 1789)

All records of this species were of beach-washed material which was found on both sides of the island. Over five years (1970, 1974, 1978, 1980 and 2000) the remains of eight birds were found with single records on six occasions, two birds on 16 January 2000 and three on 27 January 2000 and two pairs of wings being found together in 1980 [17].

### Snow Petrel, *Pagadroma nivea* (G. Forster, 1777)

Snow Petrels were recorded in August 1968 when several were reported in Buckles Bay. In July 1994 the remains of one were found at the southwestern part of the island and in November 2000 an injured bird was found on The Isthmus 17].

### Slender-billed Prion, *Pachyptila belcheri* (Mathews, 1912)

Single birds of this species were recorded five times between 1976 and 2000. In March 1976, one was caught on The Isthmus, measured and released. In May 1979, one was found in a burrow on Langdon Point sea stack and in September 1979 one was at a burrow entrance on Davis Point sea stack and on two occasions in 2000 dead birds were found on The Isthmus [14, 17].

### Westland Petrel, *Procellaria westlandica* Falla, 1946

There is only one record of this species, of a single bird close to the re-supply ship MV *Polar Queen* while it was anchoring in Buckles Bay [17].

### Short-tailed Shearwater, *Puffinus tenuirostris* (Temminck,1835)

There are seven records, from four years between 1960 and 1979, of single specimens of this species. Four records are from November, two from January and one from April, with four being beach-washed birds, one found freshly dead and one in a skua bolas. The April record was of a bird regurgitated, almost intact, by a Light-mantled Sooty Albatross, *Phoebetria palpabrata* (J. R. Forster, 1785), chick, while it was being checked on a nest, and could therefore have been collected by the parent bird some distance from the island [10, 14, 17].

### Soft-plumaged Petrel, *Pterodroma mollis* (Gould, 1844)

The remains of a Soft-plumaged Petrel were found in a Great Skua, *Stercorarius skua lonnbergi* (Mathews, 1912), territory at Saddle Point in 1979. A dead adult was also found at Saddle Point in November 1993, as well as dead birds at two places along the south coast, two at Green Gorge and one at Brothers Point. Also three burrows were found along the south coast in 1993. In October 1982 and August 2000, birds were caught during spotlighting at Green Gorge. In January 2000, a dead bird was found at Saddle Point while in November of that year a flock of 12+ was seen off North Head following the re-supply vessel MV *Polarbird* [16, 17].

### Mottled Petrel, *Pterodroma inexpectata* (J. R. Forster, 1844)

In 1956 a cranium found was identified as this species. Bourne (1981) questions this identification suggesting that it is a Soft-plumaged Petrel. One seen offshore at North Head in 1994 and a beach-washed specimen was found in Half Moon Bay in 1996 [6, 13, 17].

### Emperor Penguin, *Aptenodytes forsteri* Gray, 1844

Emperor Penguins have been recorded only twice at Macquarie Island; one bird was seen at Green Gorge on 5 March 1993 and a second at Sandy Bay on 21 February 1997. Both appeared to be juvenile birds [17].

### Adelie Penguin, *Pygoscelis adeliae* (Hambron & Jacquinot, 1841)

Three Adelie Penguins have been recorded on the island, one each in November 1950, December 1963 and January 1997. All were sighted on The Isthmus with the 1963 bird remaining at Landing Beach for three days. The 1950 bird was collected (Melbourne Museum No. B4121) [5, 17].

### Chinstrap Penguin, *Pygoscelis antarcticus* (J. R. Foster, 1781)

This species has been recorded 38 times in 23 years between 1953 and 1997. Only on one occasion were two birds seen together. Birds were recorded in all months between October and March with the most records (14) being in November. Moulting birds were noted in January, February and March. A bird banded at Dumont d'Urville on 14 February 1977 was sighted on The Nuggets beach on 23 November 1980 [6, 17].

### Snares Penguin, *Eudyptes pachyrhynchus robustus* (Oliver, 1953)

In four years between 1950 and 1979 there were seven records of this species, involving eight birds. A female, thought to have been in breeding condition, was recorded on 3 November 1957; however, egg laying is normally completed earlier than this. Immature birds were recorded moulting in February 1950, February 1977 and January 1979 [5, 6, 17].

### Erect-crested Penguin, *Eudyptes sclateri* Buller, 1888

This species was first recorded in 1912 and was reported in 21 years between 1953 and 1990 and once in 2000. Moulting birds were recorded in February and March during several years. In 1968 birds were found paired with Royal Penguins, *Eudyptes schlegeli* Finsch, 1876, one at Sandy Bay and another at Flat Creek colonies. In both cases eggs were reported but both failed. Erect-crested Penguins were seen in several years between 1963 and 1987 at the northern Royal Penguin colony, Sandy Bay [2, 6, 10, 17].

### Australasian Gannet, *Morus serrator* (G. R. Gray, 1843)

There are two records of this species at the island. In January 1977 one was seen from The Isthmus, flying just off Garden Bay. The second was of a single bird flying over The Isthmus in September 1998 [17].

### Great Cormorant, *Phalacrocorax carbo* (Linnaeus, 1758)

The first report of Great Cormorants was in March 1952, a probable sighting of two birds. In October 1957 an immature bird was seen, and in November 1957 an adult and immature bird were seen at close range. Starting in early October 1976 Great Cormorants were recorded every month until late May 1977. During this period there were 74 sightings recorded totalling approximately 300 birds, with one flock being 30+ birds. Starting again in early October 1977 and continuing monthly until March 1978 there were 18 records, totalling 40 birds with a maximum flock of nine. It is uncertain whether the break between May and October is real as there were no biologists on the island that winter and sightings may not have been logged. Undoubtedly many of the records from October 1976 to March 1978 were re-sightings of the birds/flocks. There are two other records, one in November 1979 of

a single bird out from The Nuggets and the second in January 1992 when two birds were seen at Langdon Point [6, 17].

### White-faced Heron, *Egretta novaehollandiae* (Latham, 1790)

A White-faced Heron was recorded in March 1957 and while there were several subsequent sightings that year details do not appear to have been recorded. A single bird was seen in Secluded Bay in February 1964 and there was a probable sighting of a single bird in February 1968. Sightings, from many parts of the island, occurred every month from late March to early October in 1975, with up to three birds being seen at a time. A pair of birds was seen in December 1975. There were six sightings of single birds, and one bird found dead, in 1976. In 1980 there were three records, two in 1984, one in 1985 and 1987 and two in 1998. In total 56+ White-faced Herons have been recorded at the island, occurring in all months but January [6, 17].

### Little Egret, *Egretta garzetta* (Linnaeus, 1766)

Little Egrets were recorded in 1986 and 1988. The first was a bird found dead at Hurd Point on 27 April 1986. In 1988 a single bird remained on The Isthmus for four days, 24–28 April. [17].

### Eastern Great Egret, *Ardea modesta* J.E. Gray, 1831

A bird was sighted in April 1957 which was said to be the second sighting at the island but no previous record has been found. The remains of one was taken from skuas in April 1975, and later identified in Australia, and a second dead bird was found in June that year several kilometres from the first site. In April 1986 there was a possible identification of two at Mawson Point and later that month one was seen on The Isthmus [2, 3, 6, 17].

### Intermediate Egret, *Ardea intemedia* Wagler, 1829

There are two records of Intermediate Egrets on Macquarie Island, the first being in March 1990 when one was seen on the coast just north of Green Gorge. The second was one bird recorded on The Isthmus on 9, 10 and 11 May 1990 [17].

### Cattle Egret, *Ardea ibis* Linnaeus, 1758

In August 1975 the remains of a Cattle Egret were found on The Isthmus. One was photographed feeding in Garden Bay in April 1977. Two were seen in poor condition in the Red River area on 19 April 1979, then single birds on 20 and 21 and two birds on 24 April a few kilometres south of Red River at Green Gorge. On 7 May 1979 one was found dead at Brothers Lake, a few kilometres north of Red River. One bird was recorded on The Isthmus in April 1985 and two birds were seen there in April 1986. Cattle Egrets were also recorded on 19 and 21 April 1994 at The Isthmus. During a fortnight in 1996, following a very strong easterly wind, several live birds were seen and carcasses found on the plateau. A dead bird was found on The Isthmus in April 1998. A bird sighted at Green Gorge in April 2000 was found dead there a week later. The last record is from the Caroline Cove area on 2 June of that year [17].

### Swamp Harrier, *Circus approximans* Peale, 1848

A Swamp Harrier was reported flying over North Head in October 1949. In late June 1960 and a few days later in July one was seen on the west coast around Mawson Point. In September of that year there was another sighting, also at Mawson Point, of a bird that was described as being darker than the earlier records. One was recorded in February 1967 being harassed by terns, at Mawson Point, and in September there was a possible sighting of a single bird at Mawson Point. In 1988 one was recorded in the Eagle Cave area. Both Mawson Point and Eagle Cave are along the broad raised beach-terrace of the northwestern section of the island which is covered by deep marshland associations with sections of tall tussock grass [5, 10, 17].

### Baillon's Crake, *Porzana pusilla* (Pallas, 1776)

The only record of this cryptic species is of an exhausted bird being found beside one of the buildings on The Isthmus. The bird was collected and returned to the Australian Antarctic Division for positive identification [17].

### Eurasian Coot, *Fulica atra* Linnaeus, 1758

A dead Eurasian Coot was collected in May 1957 and six others were seen frequently up to October of that year. Between June and October 1975 a single bird was recorded six times on the same pool at Green Gorge. One was recorded in September 1978 and October 1979 on a lagoon at Langdon Point and the last record is of a single bird again on the pool at Green Gorge in January 1981 [6, 17].

### Australian Pied Oystercatcher, *Haematopus longirostris* Vieillot, 1817

There are four records of the Australian Pied Oystercatcher at Macquarie Island, three in 1980 and one in 1984. In 1980 one was seen on 3, 8, and 18 of November on The Isthmus. In 1984 a single bird was recorded on the west coast around Aurora Cave [17].

### Black-winged Stilt, *Himantopus himantopus* (Linnaeus, 1758)

A Black-winged Stilt was photographed on a beach at Green Gorge on 19 April 1976. On 22 and 25 April 1976 seven birds were recorded feeding around rotting kelp on the beach at Hurd Point [17].

### Grey Plover, *Pluvialis squatarola* (Linnaeus, 1758)

In February 1984 a bird was seen below Gadget Gully, Buckles Bay. It was collected and returned to Australia where it was positively identified as a Grey Plover. The specimen is in the Melbourne Museum collection. On 5 December 1993 a single bird was sighted in Unity Bay [9, 17].

### Lesser Sand Plover, *Charadrius mongolus* Pallas, 1776

There is a single record of seven birds seen in Langdon Bay in January 1989 [17].

### Black-fronted Dotterel, *Elseyornis melanops* (Vieillot, 1818)

Two Black-fronted Dotterel were seen in September 1980 at Unity Point. A single bird was sighted at Landing Beach, The Isthmus, in October 1991 and another single bird in Windsor Bay in November 1993 [17].

### Masked Lapwing, *Vanellus miles* (Boddgert, 1783)

The first record was from September 1985 when two were seen near Duck Lagoon. In October 1992 three were recorded on The Isthmus, with a note that three had been seen on the island for several months. Three were again recorded on The Isthmus in December 1992. Two were recorded in February 1993 in

Hasselborough Bay and two on The Isthmus in December of that year. In 1994 two birds were recorded together at Langdon Point, on 27 and 30 May, and in September two were sighted at The Isthmus and one at Unity Point. There was an unrecorded number of birds at The Isthmus on 2 November 1994. One was found dead at Green Gorge in September 1998, after being seen alive several days before. Four were seen on three occasions in September 2000, twice on The Isthmus and once at Handspike Point [17].

### Latham's Snipe, *Gallinago hardwickii* (J. E. Gray, 1831)

Probable sightings of this species were reported in November 1949, on the west coast raised beach-terrace, and in November 1960 on North Head. There have been four other sightings, all of single birds. In December 1974 a bird was flushed just south of Handspike Corner, in December 1975 one was recorded at Green Gorge, in January 1980 one was sighted at Unity Point and in December 1984 one was sighted around tussock grassland on Wireless Hill. In November 1996 a snipe, species unknown, was flushed from tussock by Jessie Niccol Creek [5, 10, 17].

### Bar-tailed Godwit, *Limosa lapponica* (Linnaeus, 1758)

One bird, later identified as a Bar-tailed Godwit, was collected in October 1912 by a member of the AAE. A second Bar-tailed Godwit was collected in December 1951 [2, 4].

### Common Greenshank, *Tringa nebularia* (Gunnerus, 1767)

In 1962 a Common Greenshank was collected on the island and later deposited in the National Museum of Australia, specimen No. W 6480 [9].

### Ruddy Turnstone, *Arenaria interpres* (Linnaeus, 1758)

From 1–6 October 1971 there were several sightings of a single bird around Landing Beach on The Isthmus. In December 1976 ten were photographed at Eagle Point. Fifteen Ruddy Turnstone were recorded in Unity Bay on 5 December 1993 and on 16 January 1994 four were sighted just north of Sandy Bay. There was a record of a sighting at Unity Point in September 2000 with a note of previous sightings that year [17].

### Great Knot *Calidris tenuirostris* (Horsfield, 1821)

There is one record of this species from October 1990 when a single bird was seen by the Royal Penguin colony at Jessie Niccol Creek [17].

### Red Knot *Calidris canutus* (Linnaeus, 1758)

A specimen was collected in November 1913 by the AAE party. On 7 and 13 December 1978 three birds were seen and photographed on Midas Tarn, one of the highland lakes north of Green Gorge [2, 17].

### Red-necked Stint, *Calidris ruficollis* (Pallas, 1776)

The only record is of a single bird seen and photographed on beaches on both the eastern and western sides of The Isthmus on 3–8 October 1976 [17].

### Curlew Sandpiper, *Calidris ferruginea* (Pontoppidan, 1763)

In January 1994 two Curlew Sandpipers were recorded just north of Sandy Bay [17].

### Red-neck Phalarope, *Phalaropus lobatus* (Linnaeus, 1758)

There are five records of a single male in Half Moon Bay between 24 December 1975 and 10 January 1976. Photographs were taken on three occasions [17].

### Arctic Tern, *Sterna paradisaea* Pontoppidan, 1763)

Three records have been found of Arctic Terns at the island. Two possible sightings, one of one bird off Buckles Bay in January 1977 and one of four birds in Sandell Bay, on the west coast, in November 1991. In January 1992 two birds were photographed in Unity Bay [17].

### Pallid Cuckoo, *Cacomantis pallidus* (Latham, 1802)

One record of a bird collected at Bauer Bay on 23 October 1990 and returned to Australia for identification [17].

### Silvereye, *Zosterops lateralis* (Latham, 1802)

There is a mention of a Silvereye from 1915 but details of this record cannot be located. The only others recorded at the island are from 1994. There is a record of sightings at The Nuggets and further south at Sandy Bay on 28 May, no numbers are given in either case. Single birds were recorded below Halfway Hill on 17 September and The Isthmus on 4 October. There is also a recorded sighting, but no details, on The Isthmus on 22 November [2, 17].

### Welcome Swallow, *Hirundo neoxena* Gould, 1842

Between 1976 and 1992 there were 25 sightings of this species recorded at the island. In late March and early April 1978 single birds were seen on The Isthmus and at Sandy Bay. In the autumn of 1979 there were sightings of up to four birds at The Isthmus and other parts of the island as well as a single bird at Green Gorge. In May 1980 four immature birds were seen resting on buildings on The Isthmus and in November 1985 there are two records of birds around The Isthmus. Single or pairs of birds were recorded in the autumn of 1986, 1988 and 1992, and in the spring of 1988 and 1992 [17].

### Tree Martin, *Pterochelidon nigricans* Vieillot, 1817

There was a probable sighting of a Tree Martin on the west coast at Bauer Bay in September 1970. In October 1974 one was caught at the radio shack on The Isthmus and returned to Australia for positive identification [17].

### Common Blackbird, *Turdus merula* Linnaeus, 1758

Following unconfirmed sightings in previous years Common Blackbirds were first positively identified on Macquarie Island on 31 March 1960 with other sightings at the same site, Bauer Bay, in September of that year. In 1965 there were sightings of single birds, on The Isthmus, 6 June, and at Aerial Cove, 31 July, and records of a male and female together on 28 August and 3 September, locations unspecified. In 1965 there were 12 records, starting with the sighting of a male and one other bird south of Lusitania Bay hut on 23 April. All other sightings were of single birds, two in May, one each in June and July, three in August, one in September and three in November. The sightings were well spread around the coastal areas of the island. There was one record for each of 25 May,

30 June, 21 August and 5 September 1967 of single birds on The Isthmus, together with a sighting of a single bird at The Nuggets on 27 May. In 1970 there were sightings on the west coast, around Bauer Bay, on 23 August and 10 October and one on the east coast at Finch Creek on 2 September. single male was sighted at Caroline Cove on 21 April 1975 and a female was flushed from tussock grassland on 22 July 1976 at Hurd Point. There are two records from 1978; on 18 August a female was seen at the same site in Caroline Cove as the male in 1975 and a male was seen by Bauer Bay hut on 24 August. A single male was recorded at Sandy Bay on 18 September 1979. In 1985 there were three sightings of single birds, a male at Mt Elder on 23 July, a bird at Handspike Corner on 27 July and another in the southwest of the island on 8 August. There are four records, all of single birds, from 1990. A bird was seen at lower Finch Creek, Sandy Bay, on 24 April, another recorded on the southern end of The Isthmus on 27 August, the third, a female, on 6 September at Finch Creek and on the same day another female 3 km north at Tussock Point [10, 17].

### Song Thrush, *Turdus philomelos* C.L. Brehm, 1831

Three Song Thrushes were recorded on 30 August 1967. One was collected and the specimen is in the Melbourne Museum, No B9351. The next record is of two flushed from coastal tussock grassland north of Sandy Bay on 25 August 1976 and photographed in the same area the next day. A bird was sighted on The Isthmus in May 1977 while in 1979 there were five records, one on the east coast north of Green Gorge on 8 August, three sightings at Caroline Cove, southwestern part of the island, from 1 to 7 September and one bird again north of Green Gorge on 13 September. There were sightings of single birds at Green Gorge hut on 12 and 27 July 1985 and at Middle Beach on 23 and 26 July 1985. A record of a sighting at Handspike Point, in October 1994, does not give the number of birds. One was sighted at Secluded Beach in the winter of 1996. On 25 September 2000 there are two records of single birds a few hundred metres apart on the east coast north of Hurd Point and one several kilometres further north at Waterfall Bay. Two birds were seen in the same area at Waterfall Bay on 29 October 2000 [12, 17].

### European Goldfinch, *Carduelis carduelis* (Linnaeus, 1758)

A male goldfinch was collected on 17 April 1956. A second bird was collected on The Isthmus 5 May 1967. There were two further records both from 1977 when, on 22 May one was seen feeding in tussock grass on The Isthmus, and on 30 May one was seen flying over it [6, 11, 17].

## Rare and exotic breeding species

### Northern Mallard, *Anas platyrhynchos platyrhynchos* Linnaeus, 1758

Northern Mallard were first recorded on Macquarie Island in August 1949 when a drake was seen with 25 Pacific Black Duck, *Anas superciliosa* J.F. Gmelin, 1789, on one of the lakes. From 1964–1970 there were a further 15 sightings recorded, mainly of a single drake with Pacific Black Ducks but on two occasions in 1967 pairs were recorded. There are no further records until June 1975 and from then until the end of the year there were 29 sightings with up to 12 drakes being seen at a time and one duck with nine young. From 1976 to the end of 1980 PWS staff on the island kept more detailed records of this exotic species. Excluding 1977, when the PWS did not have staff on the island over the winter, there were a further 114 records with up to 15+ Northern Mallard being recorded at a time. In October 1979 a female was flushed off a nest with 12 eggs [5, 11, 17].

### Common Diving-Petrel, *Pelecanoides urinatrix* (J.F. Gmelin, 1789)

From 1950–1985 this species was recorded 15 times (24 birds) in 13 years. Two females collected on The Isthmus in 1956 were later identified by Bourne (1981) as South Georgian Diving-Petrels. In 1979 ten birds were measured and one bird was found incubating an egg on a sea-stack at Langdon Point. The burrows were searched again in 1988 and 1999 but no birds were found [6, 13, 14, 15, 16].

### South Georgian Diving-Petrel, *Pelecanoides georgicus* Murphy & Harper, 1916

In 1949 one specimen and in 1956 two specimens of this species were collected on the island (Melbourne Museum Nos. B 4276, M 56/8/123 and 124). The latter two were originally identified as Common Diving-Petrels. In 1993 three birds were found on eggs on Bishop and Clerk islets. In January 2000 a burrow ascribed to this species was found on a very small sea stack off the east coast [6, 13, 15, 16].

### Cape Petrel, *Daption capense* (Linnaeus, 1758)

The AAE party did not record this species. The first recorded sighting appears to be during the British, Australian, New Zealand Antarctic Research Expedition 1929–1931 (BANZARE) visit on 2 and 3 December 1930 when a small flock was sighted at the north end of the island. In 23 years between 1960 and 2000 there are 98 sightings recorded totalling over 600 birds. These range from beach-washed specimens through many sightings of single birds to flocks of over 50 on four occasions. The bulk of the birds recorded, 500+, was between August and November, with May having the only other significant numbers, 70 birds (table 1). From the 1960s it was thought that they may be breeding on offshore sea stacks but no evidence was found until March 1998 when birds were seen on nests around North Head although breeding was not confirmed. Breeding has since been confirmed [1, 2, 16, 17].

### Fairy Prion, *Pachyptila turtur* (Kuhl, 1820)

A Fairy Prion was collected in breeding condition in September 1956 and the remains of two birds were collected in October 1957. In May, September and October 1979 Fairy Prions were found in burrows on sea stacks off Langdon and Davis points. Twenty-four birds were measured and an egg found in one burrow proved that they breed on the sea stacks. Three birds on eggs and three abandoned eggs were found on Bishop and Clerk islets in the 1993 survey [6, 14, 15, 16].

### Fulmar Prion, *Pachyptila crassirostris* (Mathews, 1912)

In 1979 a female Fulmar Prion was collected on The Isthmus. The specimen is lodged in the Queen Victoria Museum, Launceston, No. 1877 1980/2/16. During bird surveys in 2000, measurements from five birds found nesting in a cave on North Head fell within the range given in HANZAB for this species. An egg was found with one of these birds in November 2000. However, either genetic analysis or

verification from an expert in prion taxonomy is required before it can be safely claimed that Fulmar Prions breed on the island [14, 17].

### Grey Petrel, *Procellaria cinerea* J.F. Gmelin, 1789

While members of the AAE did not always identify petrel and prion species they collected, often referring to them as "large or small night birds", skins in the National Museum of Australia (No. 33037) and the O'Connor collection in Wellington, New Zealand, were identified as Grey Petrel by Falla (1937). In ten of the years between 1949 and 1990 Grey Petrel were recorded 15 times, 20 birds in total, often with indications that they may be breeding on the island. During the 1990s more systematic searches were carried out, as part of a feral pest management program, and in 2000 birds were found breeding on North Head and at Green Gorge. Since then they have been recorded breeding in several other parts of the island [1, 2, 6, 10, 14, 17].

### Wilson's Storm-Petrel, *Oceanites oceanicus* (Kuhl, 1820)

Two birds were found on eggs and two birds were found together in a third burrow during the survey of Bishop and Clerk islets in 1993 [15, 16].

### Weka, *Gallirallus australis* (Sparrman, 1786)

Weka were introduced onto Macquarie Island, from New Zealand, around 1870, although there could have been an earlier introduction in the 1840s (Cumpston 1968). By the early 1900s they were common and widespread around the island and often caught for food by the people working on the penguin oil industry and AAE members. When a rabbit control program took effect in the early 1980s Weka numbers dropped rapidly, most likely due to feral cats using them to replace rabbits as a food source. Starting in 1985 the PWS began a program to eradicate the Weka from the island and the last recorded was one shot at Bauer Bay in December 1989 [2, 17].

### Common Redpoll, *Acanthis flammea* (Linnaeus, 1758)

Common Redpoll were first recorded on Macquarie Island by the AAE party on 8 February 1912 when seven birds, of which three were collected, were seen feeding at Sandy Bay. Others were seen at Lusitania Bay and a nest found during the AAE stay. None were seen during the BANZARE visit in 1930. This exotic species is now well established and small flocks can be seen in most parts of the island, from sea level to higher parts of the plateau. Nests have been found in clumps of fern, tall tussock grass and the flower heads of the Macquarie Island cabbage, *Stilbocarpa polaris* A. Gray [2, 17].

### Common Starling, *Sturnus vulgaris* Linnaeus, 1758

This species was first positively recorded at Macquarie Island on 2 and 3 December 1930, during the BANZARE visit, when pairs were seen on five occasions. They breed in small groups on rock stacks and cliffs around the island. In winter they form one or more flocks of several hundreds of birds and can be seen feeding on the plateau and in unoccupied penguin colonies [2, 17].

## Possible identifications

### Double-banded Dotterel, *Charadrius bicinctus* Jardine & Selby, 1827

A bird tentatively identified as a Double-banded Dotterel was seen on The Isthmus in October 1978 [17].

### Hooded Plover, *Thinornis rubricollis* (J.F. Gmelin, 1789)

One possible sighting of four birds on the reef at the northern end of Sandy Bay in January 1981 [17].

### Little Tern, *Sternula albifrons* (Pallas, 1764)

There is one record of a possible sighting at Bauer Bay, west coast, in December 1975 [17].

### Australasian Pipit, *Anthus novaeseelandiae* (J.F. Gmelin, 1789)

This is a tentative identification of a bird seen on The Isthmus in July 1978. In 1956 and 1957 "pipits" were also sighted but could not be specifically identified [6, 17].

## DISCUSSION

In considering the recorded sightings and the numbers of birds recorded (tables 1 and 2) several factors need to be taken into account. While the records cover over 54 years, during that time there have not always been competent bird observers on the island throughout parts or the whole of every year. The majority of ANARE personnel are based at the Station, on The Isthmus, and through winter periods there were often few observers away from there. Some parts of the island may not be visited for months at a time, or even years along sections of the west coast. The shoreline habitat in the vicinity of Langdon Point and Unity Point provides the most extensive and sheltered environment for shorebirds on the island. It is not surprising therefore that there are numerous records of at least four waders from this seldom-visited locality and this habitat may be even more regularly frequented than is revealed by the records so far. In many cases the number of birds recorded, either monthly or over several months, could be multiple sightings of the same bird/s. In most cases it is therefore not possible to tell whether apparent changes in the occurrence of a species at the island are real or due to observational/recording effort.

From records covering the period 1901–2000 it has been possible to identify 56 taxa of non-breeding birds and 11 species of rare, exotic or introduced breeding birds at Macquarie Island (tables 1 and 2), excluding the 19 regular breeding species. In addition there are also four species whose identity is doubtful (table 3). Of the 56 non-breeding species 20 are pelagic seabirds and four are exotic species introduced into the Australasian region. The breeding species recorded here consist of seven seabirds, three exotics and one introduced species. The seven breeding seabird species are known to, or likely to, have bred previously on the main island and four appear to be re-establishing there, as control and eradication programs for introduced species, started in the late 1970s, take effect (Brothers & Copson 1988, Copson 2004, Brothers & Bone 2008).

Of the 66 positively identified species 14 are holarctic breeding species which migrate to the southern hemisphere during the northern winter period. Seven of these14 species

have been recorded on other sub-Antarctic islands. Some, such as the Cattle Egret, appear to be extending their range generally (Crosby 1972, Prince & Croxall 1983, Trivelpiece *et al.* 1987).

Thirty-four out of 65 identified species, excluding the introduced flightless Weka, could not, or are unlikely to, rest on the open ocean and must therefore have travelled non-stop from their point of departure to Macquarie Island. The influence on the dispersal of birds of the extreme winds that can occur in the region has been discussed in several papers (Williams 1953, Merilees 1971b, Adamson *et al.* 1988, Bridgman & Maddock 1994, Brothers & Bone 2008). This is well-illustrated with the racing pigeon (Rock Dove above) which was released in Victoria, Australia, on the morning of 29 October 1989 in a race to northern Tasmania. It was caught at the ANARE Station at 10 am on 31 October having covered a distance of at least 2000 km, an unlikely great circle route (PWS unpubl. records). Over this period there had been a strong northwesterly air-stream extending from northern Victoria. The wind at Macquarie Island had increased from a light average wind speed of 7.1 km/h, maximum gust 48 km/h W, on 29 October, through 24.5 km/h average, maximum gust 67 km/h NNW, on 30 October, to 43.4 km/h average, maximum gust 96 km/h NNW, on 31 October. The wind run, i.e., the amount of wind passing a fixed point in 24 hours, recorded at the Station up to 9 am on 31 October was 1042 km. The bird appeared in good condition when found.

There are several interesting but unverifiable observations from the earlier literature. In February 1903 the sealers caught at The Nuggets, and ate, what they described as a "white swan" (Cumpston 1968). On 26 May 1913 Anderson, one of the AAE party, reported seeing a "... thrush type bird again at Catch Me Point", (Hamilton 1912–13); was this the first sighting of the Song Thrush or Common Blackbird? Possibly the first sighting of the Common Starling on the island occurred on 8 September 1913 when "... two small birds like starlings" were seen between Sandy Bay and the west coast" (Hamilton 1912–13). In addition in 1983 a glass container, possibly an old food jar, was found in the ruins of the sealers' hut at The Nuggets with a bird preserved in it. The specimen appeared in good condition as the container had a sound cork stopper and was still full of fluid. It was later sent to the Queen Victoria Museum, Launceston, where the cleaned specimen was identified as an immature Black-faced Monarch, *Monarcha melanopsis* (Vieillot, 1818). It was placed in fresh alcohol and is registered as specimen1985/2/150. The specimen and container were unlabelled but one possible source was the taxidermist Joseph Burton who worked at The Nuggets, in the penguin oil industry, from 1896–1900 and was permitted to collect specimens in his spare time (Cumpston 1968). If so, it is from before the period under review.

An exotic species management program has been conducted at Macquarie Island since the late 1960s which, together with climate changes being recorded in these regions, will almost certainly result in changes to the status of many avian species recorded at the island. While detailed scientific studies will be necessary to fully evaluate such changes the contribution of long-term observations and record keeping, whether incidental to the main study, or by scientists working in other fields, keen amateurs or just casual observers, should not be overlooked.

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# THE AVIFAUNA OF BISHOP AND CLERK ISLETS AND ITS RELATIONSHIP TO NEARBY MACQUARIE ISLAND

by Nigel Brothers & Rod Ledingham

(with one table)

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A first comprehensive survey of seabirds at Bishop and Clerk islets conducted on 23 December 1993 recorded a total of 12 species. During a three-hour-long visit by helicopter, nine species were found breeding, mostly in nests on the ground, but also in burrows dug in shallow soil. These included the largest known colony of Black-browed Albatross, *Thalassarche melanophris*, in Australia. Ten bird species are now known to breed at the islets. Species accounts are given including data on morphometrics, abundance, habitats, breeding, threats, interspecific competition for space and unpublished information.

**Key Words: sub-Antarctic, Bishop and Clerk islets, Macquarie Island, seabird abundance, island management.**

## INTRODUCTION

Bishop and Clerk islets are a tiny cluster of near-barren rocks projecting from a submarine ridge 33 km off the south coast of Macquarie Island in the south Pacific Ocean (54°30'S, 159°E). Politically, this region is part of Tasmania, a state of Australia, therefore the status of species occurring there is subject to both Australian federal and state conservation processes. Lying within the Macquarie Island Marine Park boundary, these islets have been assigned the status of a Special Management Area (SMA) as part of the Macquarie Island Nature Reserve and World Heritage Area, recognising that, whilst the wildlife there is of scientific interest "the risks of adverse impacts outweigh almost any justification for visiting them" (Parks and Wildlife Service (PWS) 2003: 87).

The islands were visited in 1965 (MacKenzie 1968), and in 1976 (Lugg *et al.* 1978) prior to this declaration of importance. The timing of these earlier visits in relation to seabird breeding activity was not ideal or comprehensive. Of particular importance was to ascertain how a number of petrel species were managing to maintain breeding populations on rock stacks immediately adjacent to Macquarie Island's coastline, despite apparent successive total breeding failures (Brothers 1984). Were the Macquarie Island populations being supplemented by immigration from elsewhere and were Bishop and Clerk islets the likely source as inferred by Lugg *et al.* (1978)? Alternately, were there other, as yet unknown, processes enabling these petrel colonies to persist? The implications of immigration from offshore populations for petrel population recovery following vertebrate pest management on Macquarie Island (Copson 1995) were considered to be of potential significance. Because of this and also to ensure efforts to update and review the regional status of albatross (Gales 1993, 1998) included information on numbers known to breed on Bishop and Clerk islets, a visit was planned. Attempts were unsuccessful in 1982 and 1988, but a visit was successfully accomplished in 1993. This paper provides the only comprehensive avifauna account for the locality from what may be the last visit permitted.

## MATERIALS AND METHODS

One helicopter, accompanied by the Antarctic resupply vessel MV *Icebird* was used to deploy a field party onto Bishop Islet the largest islet in the group and the only one habitable to wildlife for breeding, on 23 December 1993. To maximise the benefit of what could only be a brief three-hour (1230–1530 hrs) visit, a total of seven personnel participated in various work including studies on invertebrates (Davies *et al.* (1997)), seals (Goldsworthy unpubl. data) and Macquarie Shag, *Leucocarbo atriceps purpurascens* Brandt, 1837, diet (Kato unpubl. data) as well as this census of birds.

All habitats were thoroughly searched and direct counts made of bird species abundance and their breeding status noted. The relative importance of the seabird populations inhabiting Bishop and Clerk islets was then assessed against the most appropriate data available for nearby Macquarie Island.

Whilst at the time of this visit no access restrictions specific to the locality applied, as they do today, specific quarantine precautions were taken to preclude the inadvertent or uncontained transfer of organisms between Macquarie and Bishop and Clerk islets.

## SPECIES ACCOUNTS

Of the 12 species of seabirds observed, nine were confirmed breeding, five of these being first records for this locality and two being new records for the Macquarie Island region or not known as species breeding in Australia (excluding the Australian external territories of Heard and McDonnell islands). One species, which had previously been recorded breeding here, was present but not breeding, with the result that ten species are now known to breed at this locality.

## Wilsons Storm-Petrel, *Oceanites oceanicus* (Kuhl, 1820)

Three burrows, two containing an adult and one egg, and one containing two adults with no egg were found in the same vicinity and habitat as the diving-petrels. Prior to the destruction caused by Black Rats, *Rattus rattus* (Linnaeus, 1758), European Rabbits, *Oryctolagus cuniculus* (Linnaeus, 1758), Feral Cats, *Felis catus* Linnaeus, 1758 and Weka, *Gallirallus australis* (Sparrman, 1786), it is likely that this species was abundant on Macquarie Island itself. Its presence on Bishop Islet is the first record of the species in this region and a new breeding record for Australia (not including its external territories of Heard and McDonnell Islands plus those in the Australian Antarctic Territory (Marchant & Higgins 1990)).

Breeding by this species on Bishop Islet is added confirmation that rats do not exist here despite perhaps having had at least one opportunity from shipwreck to do so (Cumpston 1968). Regardless, evidence from other islets adjacent to Macquarie Island suggests that even if rats did come ashore at any time on Bishop Islet they could not survive long-term under the environmental conditions here (Brothers & Bone 2008).

One egg measured 32.9 x 24.1 mm. Measurements from three adult birds are given in table 1.

## Black-browed Albatross, *Thalassarche melanophris* (Temminck, 1828)

A total of 141 nests was recorded. Of these, 78 contained a chick, 13 an egg, ten had egg-shell fragments and 40 were empty. Despite a general lack of nest-building material all nests were well-formed suggesting that they had all been constructed and occupied that season. Unoccupied nests are soon destroyed by neighbouring birds, particularly at such localities where construction material is scarce.

Each nest built represents a potential breeding pair and, although it is likely that an egg was not actually laid in every nest (at Macquarie Island colony 83% of nests have eggs laid in them (DPIWE unpubl. data)), 140 breeding pairs is considered a realistic population estimate. This estimate is the same as that assigned to this population by Garnett & Crowley (2000).

The data indicate a reduced breeding success of about 35% at this early stage of the breeding cycle. This is consistent with observations at the Macquarie Island colony (DPIWE unpubl. data).

**TABLE 1**
**Standard measurements (in mm) of three adult Wilson's Storm-Petrels on Bishop Islet**

| | Bird 1 | Bird 2 | Bird 3 |
|---|---|---|---|
| Culmen | 12.9 | 12.8 | 14.0 |
| Bill width at base | 6.9 | 6.9 | 6.8 |
| Tarsus | 35.8 | 33.7 | 33.5 |
| Mid toe + claw | 29.9 | 26.8 | 28.1 |
| Mid claw | 4.5 | 4.5 | 5.4 |
| Wing | 159 | 148 | 149 |
| Tail | 65 | 57 | 60.5 |

Nesting habitat differences occur between the two localities and could account for any differences in breeding success. Nest site exposure to adverse weather conditions at Bishop Islet, the availability of an entirely rock substrate with limited nest-building material together with a high proportion of sites being relatively difficult for adults to access may account for differences in breeding success between the two sites.

Interchange between sites is supported by the observation that one 11-year-old Macquarie Island bird banded as a fledgling was present in the Bishop Islet colony. Given that the bird has never been recorded back in the Macquarie Island colony over the 20-year period, despite comprehensive searches (DPIWE unpubl.) it is suggested that it may have permanently established in the Bishop Islet colony. Although fledgling survival seems to be significantly higher at Macquarie Island compared to populations elsewhere (Terauds *et al.* 2005), recruitment into the Bishop Islet colony cannot be discounted. The annual incidence of unbanded birds being encountered in the Macquarie Island population (which consists entirely of banded birds), is less than 5% (Terauds pers. comm.) suggesting immigration from Bishop Islet (where no birds have been banded) to be a rare event.

With as few as 45 breeding pairs now at just one site on Macquarie Island (PWS 2003), the colony on Bishop Islet has been recognised as significant for the region since MacKenzie's first observations (1965). The timing of the two previous visits to this site together with uncertainties about the completeness of the counts undertaken, precludes comparison or comment on population trends.

Emigration from the Macquarie colony is likely to be influenced by the fact that the greatest numbers of individuals present in the region with which to interact socially at sea will most likely be from the Bishop Islet colony. The aggregation of birds created by fishing vessel activity, especially in the vicinity of the island, is likely to increase the potential for this to occur.

Other potential threats to the stability of the Bishop Islet colony include the observation that both Rockhopper and Royal penguins access the Bishop Islet colony, breeding adjacent to the albatross in similar habitat. In the same vicinity, penguins appear to occupy the more suitable sites, suggesting dominance, which could have a negative impact on albatross population size should penguins increase in abundance. As it is, this occupation of more suitable nesting habitat by the penguins might be lowering albatross breeding success. Direct nest site competition such as this is not a factor influencing performance at the Macquarie Island Black-browed Albatross colony.

## Salvin's Albatross, *Thalassarche cauta salvini* (Rothschild, 1893)

One individual of adult appearance was present amongst nesting Black-browed Albatross. Whilst it gave no indication of having an association with any specific site or nest, the possibility that this species does nest here cannot be discounted. A new colony of this species has recently become established in the Crozet Archipelago (Jouventin 1990) and such an establishment on Bishop Islet could disrupt or displace less aggressive Black-browed Albatross. There are no previous records of Salvin's Albatross ashore but small numbers are sighted regularly at sea in the Macquarie Island region accompanying a fishing vessel (Australian Fisheries Management Authority (AFMA), unpubl. fisheries observer

data). It is not uncommon however for an individual albatross to frequent the breeding colonies of a different albatross species but never to breed at that locality.

### Fairy Prion, *Pachyptila turtur* (Kuhl, 1820)

MacKenzie (1968) and Lugg *et al.* (1978) hypothesised that Fairy Prions could nest in rock crevices on Bishop Islet. This hypothesis was confirmed with birds found occupying two cliff crevices within 20 m of the summit. Three birds were incubating eggs and there were another three unattended eggs. Both crevices provided very limited nesting opportunities and whilst one or two other suitable crevices may have been overlooked, this locality can support only very small numbers of this species. The occupied crevices here are reminiscent of the other sites adjacent to Macquarie Island at which this species nests (Brothers 1984).

Despite being small, the Bishop Islet colony is significant for the region because only small numbers (about 40 pairs) are known to occur at two localities elsewhere and, unlike at Bishop Islet, all are vulnerable to the predatory Black Rat.

### Common Diving-Petrel, *Pelecanoides urinatrix* (J.F. Gmelin, 1789)

One burrow, containing a well-developed chick was found beneath *Colobanthus* cushions adjacent to the summit of Bishop Islet. It is estimated that this chick would have originated from an egg laid in late September, a timetable consistent for the species. No other burrows of this species were found to occur here. As with the Fairy Prion, this species has been all but exterminated on Macquarie Island by introduced predators and survives at only one, or possibly two, other offshore sites adjacent to Macquarie Island but in very low numbers (Brothers 1984, Brothers & Bone 2008).

### South Georgian Diving-Petrel, *Pelecanoides georgicus* Murphy & Harper, 1916

Three burrows found adjacent to the summit of Bishop Islet, beneath *Colobanthus* cushions, each contained an adult bird incubating an egg. Whilst Murphy & Harper (1921) speculated that this species breeds in the Macquarie Island region this is the first confirmation of this and also constitutes a new breeding record of the species for Australia, not including the external territories of Heard and McDonald Islands (Marchant & Higgins 1990).

### Gentoo Penguin, *Pygoscelis papua* (J.R. Forster, 1781)

No Gentoo Penguins were found breeding on Bishop or Clerk islets, but 149 birds were recorded ashore with eight Royal Penguins (*Eudyptes schlegeli*) on a smaller islet just to the east of Bishop Islet. This unnamed islet offered easy access for these species. Given that birds forage daily from Macquarie Island (Hindell 1989), it is likely that these sites are used regularly.

It has been hypothesised that Gentoo Penguins compete directly for food resources with Macquarie Shags (Brothers 1985), so maximising competition for these resources at this distant locality might be assisted by having access to rest ashore as an option instead of swimming 33 km directly back to Macquarie Island.

This is uncharacteristic behaviour, but the data presented here, indicate that such use of sites does occur.

Although Gentoo Penguins, together with two other penguin species, the Rockhopper Penguin *Eudyptes chrysocome* and the Royal Penguin *Eudyptes schlegeli* could access Bishop Islet itself, typical nesting requirements of the 3800 pairs on Macquarie Island (Robinson in PWS 2003) are lacking. These observations together with this more appropriately-timed visit to ascertain breeding timetables, removes earlier uncertainty (Lugg *et al.* 1978) about breeding status here.

### Rockhopper Penguin, *Eudyptes chrysocome* (J.R. Forster, 1781)

Rockhopper Penguins occupied 166 nest sites in typical rocky habitat adjacent to, and interspersed with, Royal Penguins.

Although previously known to breed here (Lugg *et al.* 1978), this is the only reliable population estimate.

Approximately 100 000 pairs of Rockhopper Penguins breed on Macquarie Island (Rounsevell & Brothers 1984) and similar interactions with Royal Penguins are encountered there.

### Royal Penguin, *Eudyptes schlegeli* Finsch, 1876

Nesting on Bishop Islet was recorded with 457 occupied nests encountered. Eight of these had one egg and the rest contained chicks. A vast 850 000 pairs breed on Macquarie Island (Copson & Rounsevell 1987). Although only eight individuals were ashore with Gentoo Penguins on the small islet to the east of Bishop Islet the earlier accounts of Mackenzie (1968) indicate high usage of this site at times but not for breeding purposes.

### Macquarie Shag, *Leucocarbo atriceps purpurascens* Brandt, 1837

Three localities at which this species nest were found on Bishop Islet with a total of 164 nests, 148 at the main site to the east, 15 in the southeast and a single one in the northwest amongst nesting Black-browed Albatross.

There were 59 empty nests, four with one egg, nine with two eggs and four with three eggs plus 22 with one chick, 59 with two chicks, three with three chicks and two with one egg and two chicks and one with two chicks alive, one dead. Because nest site retention by adults in this species is very strong and persistent (Brothers 1985), the above count is considered an accurate indication of total breeding pairs here.

If nest content at this time of year and in this season is any indication (in particular the proportion of nests that were either empty or contained only one chick) productivity appears to be less than at a Macquarie Island colony studied (Brothers 1985). Explanations for this may include food shortages, a factor believed to influence breeding success of this species (Brothers 1985) even where birds have access to areas that remain suitable for foraging during inclement weather. In addition to limited food availability, the necessity of reliance upon only seaweed for nest building here may also diminish breeding success. Unlike nests on Macquarie Island that are all constructed from readily available terrestrial plant species (Brothers 1985) with only the odd shore-cast seaweed fragment incorporated, Bishop Islet nests are constructed

exclusively of seaweed collected by the birds at sea. The only terrestrial plant, *Colobanthus muscoides* Hook.f., on Bishop Islet is not sufficiently abundant to sustain its use for nest building nor is it really suitable for this purpose. A shortage of nest material will exacerbate the King Shag's destructive habit of robbing nest material and would have a significant impact on the breeding success of this colony.

MacKenzie (1968) considered that the Bishop Islet colony could act as a reservoir for maintaining shag numbers at Macquarie Island colonies where nest sites of lower elevation are more susceptible to storms. Whilst it is true that Macquarie Island colonies can be susceptible to storm flooding (Brothers 1985) it is most likely that the Bishop Islet portion of the population was overestimated and factors affecting productivity on Bishop Islet had not been considered by MacKenzie in reaching this conclusion. Furthermore, given the colony tenacity evident in Macquarie Island colonies (Brothers 1985), it is unlikely that Bishop Islet breeding birds and their offspring would enter the Macquarie Island breeding population.

Variation in annual individual colony counts on Macquarie Island does suggest, however, some movement between colonies. To illustrate this point, counts at the largest of all colonies were 311 nests (in 1975), 332 (1976), 348 (1977), 343 (1988), 364 (1993) and 417 (1999). The next largest colony varied from 51 nests to 132 nests over this period. Whilst nesting sites are generally used every year, some colonies can be abandoned for a season or more (Brothers 1985). This perhaps accounts for much of the colony nest count fluctuations. Regardless, the tendency for large colonies to remain large and, if not at the same exact site, always in the same region supports the belief that similar numbers are likely to nest at Bishop Islet each year.

Limited benthic foraging areas, upon which this species appears to be dependent (Brothers 1985), exist in the vicinity of Bishop Islet. Prevailing weather conditions and distance largely prevent Bishop Islet birds and Macquarie Island birds from exploiting each another's foraging grounds. Foraging area is also likely to determine population size on Bishop Islet.

Given the fluctuations in nest counts experienced on Macquarie Island over the years, the validity of using a single count from Bishop Islet to provide a total species population estimate should be treated with caution. With 164 nests, the Bishop Islet colony is significant, being similar in size to the second largest colony on Macquarie Island but three or more times larger than all others.

A previous estimate of population size of King Shags for the Macquarie Island area of 760 pairs (Brothers 1985) assigned only 100 pairs to the Bishop Islet colony. Combining the one accurate count of 164 nests with the most recent Macquarie Island census in the 1999/2000 breeding season indicates a total population size for the species of 1047 pairs.

### Great Skua, *Stercorarius antarctica lonnbergi* (Matthews, 1912)

Two adult birds and an empty nest were encountered which is consistent with observation made of this species during previous visits. There are approximately 200 pairs of this species breeding on Macquarie Island (Skira 1984).

### Kelp Gull, *Larus dominicanus* Lichtenstein, 1823

Although six adults and 13 juveniles were present, no evidence of breeding was found. Confirmation that this species does breed here was made by MacKenzie (1968) and it is considered likely that at least one pair does so each year.

At nesting sites of this species on Macquarie Island a breeding pair is invariably assisted in site defence by several other adult birds as well as juveniles. Therefore the numbers observed on Bishop Islet do not necessarily indicate that more than one pair may breed. The total population for the Macquarie Island region is 127 breeding pairs (Brothers & Bone 2008).

## DISCUSSION

Although four species of burrow-nesting petrels were found nesting on Bishop Islet for the first time, absence of Blue Petrels, *Halobaena caerulea* (J.F. Gmelin, 1879), was unexpected. Because suitable habitat is scarce it is not possible for even small numbers of this species or for larger populations of the other species actually found, to exist here. Not only does habitat availability limit abundance, its characteristics also dictate that only the smaller burrow-nesting petrel species can occupy this site.

The species that were found have provided further evidence of which species may have once constituted the burrow-nesting avifauna of Macquarie Island prior to extermination by vertebrate pests. This may also be indicative of species that are expected to re-occupy Macquarie Island after completion of the Integrated Vertebrate Pest Management Plan (Copson 2004). A notable exception to this hypothesis is the absence of Cape Petrels, *Daption capense* (Latham, 1758), on Bishop Islet despite suitable nesting habitat, which suggests this species was not eliminated from Macquarie Island by introduced pests. Cape Petrels however, do frequent Anchor Rock, an apparently less suitable site adjacent to Macquarie Island (Lugg *et al.* 1978, Brothers & Bone 2008) so could exploit more favourable nesting conditions, should these be created.

There are insufficient numbers of all species on Bishop Islet to affect the existing conservation status that has been assigned to them by Garnett & Crowley (2000).

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# THE RESPONSE OF BURROW-NESTING PETRELS AND OTHER VULNERABLE BIRD SPECIES TO VERTEBRATE PEST MANAGEMENT AND CLIMATE CHANGE ON SUB-ANTARCTIC MACQUARIE ISLAND

by Nigel Brothers & Catherine Bone

(with nine text-figures and 17 tables)

**Brothers, N. & Bone, C.** 2008 (31:x) The response of burrow-nesting petrels and other vulnerable bird species to vertebrate pest management and climate change on sub-Antarctic Macquarie Island. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 123–148. ISSN 0080-4703. Department of Primary Industries, Water and Environment, GPO Box 44, Hobart, Tasmania 7001, Australia (NB), 178 South Arm Drive, Wonga Beach, Queensland 4873, Australia (NB*, CB) *Author and address for correspondence. Email: brothersbone@yahoo.com.au

Pest species management is causing rapid and significant changes to burrow-nesting petrel populations on sub-Antarctic Macquarie Island. The Weka, *Gallirallus australis*, was eliminated by 1989 and the Feral Cat, *Felis catus*, eradicated in 2000. The most abundant burrow-nesting petrel species currently, White-headed Petrels, *Pterodroma lessonii*, Antarctic Prions, *Pachyptila desolata*, and Sooty Shearwaters, *Puffinus griseus*, have yet to increase in numbers, but are expected to do so in the absence of cats. This study found evidence that Grey Petrels, *Procellaria cinerea*, began breeding again on the island in 1999, after an absence of over 100 years. Blue Petrels, *Halobaena caerulea*, and Fairy Prions, *Pachyptila turtur*, were found to be re-colonising Macquarie Island from offshore stacks after a similar absence. South Georgian Diving-Petrels, *Pelecanoides georgicus*, were also possibly recolonising the island. Despite the presence of Black Rats, *Rattus rattus*, most of the bird species discussed are considered capable of population increase. If European Rabbits, *Oryctolagus cuniculus*, are not eliminated or maintained in reduced numbers, some petrel populations will never fully recover. Climate change could have a negative impact on burrow-nesting petrels, and is likely to exacerbate the detrimental effects of the remaining pest species on vulnerable indigenous bird species, compounding the need for remedial action against rabbits in particular. Together with predictions that other petrel species will now return to breed, certain terrestrial bird species, alien to the region, may invade Macquarie Island as a consequence of the combination of pest eradication and changing climatic conditions.

**Key Words: Burrow-nesting petrel, pest eradication, climate change, population response, Macquarie Island.**

## INTRODUCTION

The status of burrow-nesting petrels on Australia's sub-Antarctic Macquarie Island situated at 54°30'S, 158°56'E, was reported by Brothers (1984). However, this was a period when vertebrate pest management effort directed at the European Rabbit, *Oryctolagus cuniculus* (Linnaeus, 1758), Feral Cat, *Felis catus* Linnaeus, 1758, Black Rat, *Rattus rattus* (Linnaeus, 1758), House Mouse, *Mus musculus* Linnaeus, 1758 and Weka, *Gallirallus australis* (Sparrman, 1786) had either made little impact, or had not yet commenced. In the following period up to 2000, the rabbit population was dramatically reduced (Brothers *et al.* 1982) and maintained at a much lower level (Copson 2004), Weka were eliminated by 1989 (Copson 1995) and cats by 2000 (Copson 2002). Because these events are likely to precipitate rapid changes in the distribution, abundance and species composition of the island's avifauna, an update of the population status of monitored affected species is presented here. Further threatened species management outcomes can be assessed against this baseline information (Copson 2002, 2004, Parks and Wildlife Service 2006, Threatened Species Section 2007).

During the course of burrow-nesting petrel monitoring, the consequences of vertebrate pest management, aspects of which were the subject of earlier extensive discussion (Brothers & Copson 1988, Scott 1988, Copson 1995, Copson & Whinam 1998, 2001), have become complicated by the apparent impact of recent specific climatic events and trends. Tweedie & Bergstrom (2000) discussed some of the impacts of changing climatic conditions on Macquarie Island and Chown & Smith (1993) discussed similar impacts for the sub-Antarctic Prince Edward Islands. Already a rapid increase in temperatures over the past 50 years together with trends in other climatic features have been documented by Adamson *et al.* (1988) and Pendlebury & Barnes-Keoghan (2007) for Macquarie Island. Specific observations of a climatic nature during the period of the study reported here are discussed in an attempt to ascertain the distinction between impacts of climate changes on indigenous fauna and impacts of vertebrate pest management actions alone or in combination.

The status and population trends of ten burrow-nesting seabird species from 1975–1999 are provided here. These data are examined in the context of five vertebrate pests that have invaded the island with particular emphasis on the three species (rabbits, rats and mice) remaining on the island after 2000. The effects of climate change and likely impacts on indigenous bird species and feral species are examined and discussed.

## MATERIALS AND METHODS

After the initial assessment of distribution and abundance of burrow-nesting petrels by Brothers (1984), further assessments were made in 1988 (28 October–2 December), 1993 (28 October–24 December) and in 1999 (11 November–11 February 2000). This most recent assessment was the last breeding season for the majority of seabird species during which cats were known to be present (Copson 2002). The census method followed that described in Brothers (1984),

being tailored for the various species to suit logistics and maximise data accuracy in different habitat types, and with consideration given to the possibility that any future monitoring may be done by individuals with varying degrees of expertise. An additional factor influencing the census method design was the probability that an altered population density could change the logistics and reliability of any future census. For example, while current nesting-densities may permit efficient and accurate counting, this ability may be lost in the event of a substantial increase in nesting density and/or in the event of rabbit eradication, with the reversion of heavily rabbit-grazed habitat to ungrazed, densely vegetated habitat, which is more difficult to search. The potential of such factors to compromise the accuracy and, therefore, comparability of counts over time, determined for example, that transects were kept narrow but not necessarily short. As population monitoring was the principal objective, count areas or transect site selection was biased to localities where the different species still remained, rather than in the original main nesting areas now no longer occupied due either directly (predation) or indirectly (nesting habitat modification) to one or more of the vertebrate pest species.

For consistency, localities of transects and records of breeding colonies, followed Brothers (1984), using a 1 km$^2$ grid system (and tenths thereof) based on the intercepts of 54°48'S, 158°45'E from the 1:50000 map produced by the Division of National Mapping, Canberra (fig. 1). It should be noted that this plotting method was replaced in 2000, after the island's actual geographical position was corrected (G. Copson pers. comm.).

Species population trends were measured by means of repeated counts of either established permanently marked count areas (tables 1–4) or counts of discreet colonies identified by their locality name and six-figure grid reference point (tables 5,6). This latter method was used for Sooty Shearwaters, *Puffinus griseus* (J.F. Gmelin, 1789), a widespread species in discreet, generally small colonies with island-wide scattered colonies that are the most accessible to all potential predators. Thus, Sooty Shearwaters were considered to be the best potential overall indicator of pest and pest-management impacts on petrels during this period. A sub-sample only of all Sooty Shearwater colonies was used to indicate population trends, because of uncertainty as to whether some newly-discovered colonies were new or overlooked previously in terrain and habitat that readily conceals burrows. Population trends were calculated from the difference between each separate year when counts were done. Consistent with the methods described by Brothers (1984), a burrow that showed signs of recent occupation was considered indicative of one potential breeding pair.

The first permanently-marked population monitoring sites were established in 1988 and were primarily for Antarctic Prions, *Pachytila desolata* (J.F. Gmelin, 1789), and White-headed Petrels, *Pterodroma lessonii* (Garnot, 1826). For prions, two 100 m x 200 m (fig.1) rabbit population monitoring count areas (Copson *et al.* 1981) were initially selected because not only were prions known to breed within them, but because any relationship between prion and rabbit abundance was likely to become apparent, as rabbit counts have been maintained at these sites for over 25 years (Copson 2004). These count areas, one at Prion Lake, the other at South Square Lake, had their boundaries marked with plastic pipe. Burrows were counted 5 m each side of a line between all corner markers as well as dissecting across from the centre of each side of the count

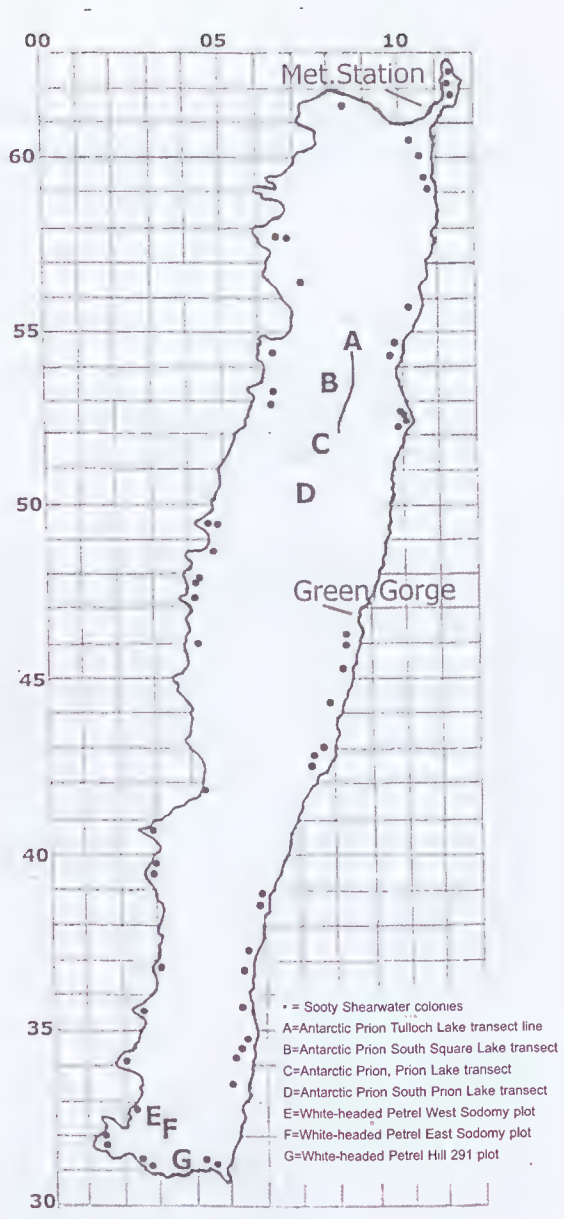


*FIG. 1 — Macquarie Island with the 1 km$^2$ numbered grid system for colony site identification, the location of petrel monitoring sites, Sooty Shearwater colonies, and the Green Gorge and Meteorological Station rainfall observation sites.*

area. South Prion Lake monitoring site (fig. 1, table 1), a straight 250 m x 10 m transect with a mid-line that had a plastic pipe marker at 50-m intervals, was not established until 1999. The track section between the outflow of Tulloch Lake and the Bauer Bay to Sandy Bay track intersection ("Fourways"), a distance of approximately 2 km, was chosen in 1988 to monitor changing prion abundance. This area was considered to have abundant suitable but as yet little occupied habitat. Transect width was 2.5 m and situated on each side of the track.

Counts at prion monitoring sites were undertaken between late November and early December in each year, the period of greatest burrowing activity. For the count years of 1993 and 1999, prion call playback tapes were used in an attempt to improve burrow occupancy counts, ascertain the extent of multiple pairs using one burrow entrance and determine the

extent to which rabbit burrows on transects were occupied by prions. Although prions are known to frequently exploit rabbit burrows for nesting (Brothers 1984) the extent of this is not easily ascertained. Therefore, counts were maintained of various burrow types from which a response was generated by call playback (table 1). Each burrow entrance was subjected to call playback for 5 seconds with negative and positive responses noted (table 1).

The first permanently-marked White-headed Petrel monitoring plots were established in 1993. One at 031312, in herbfield habitat, extending 50 m downhill to the east of Hill 291 summit (fig. 1), 50 m across the slope northwards with the third boundary then running back to Hill 291 summit, forming a triangle. This area for many years (at least since 1975) has been denuded of *Poa foliosa* (Hook. f.) tussock by rabbit grazing. The other, a rectangular plot in un-grazed tussock at the western end of Sodomy Ridge (023326) marked with plastic pipes, extends east-west for 60 m and is 5 m in width. An additional White-headed Petrel monitoring plot (approximately 20 $m^2$) was established in 1999 further east on Sodomy Ridge (025326) and marked with plastic pipes. At this time dense tussock covered the northern half of the plot, but the southern half had been partially destroyed by rabbit grazing, with adjacent vegetation being heavily impacted. Counts of White-headed Petrels were made through the last half of November into December with active burrows remaining obvious into January.

For other species populations, only estimates are available. These were based on the extent and abundance of nesting sites. More accurate counts were not possible due to the duration and timing of visits, the nature of nesting habitat, or due to the extent of interconnected burrows characteristic of the colonies on offshore stacks. Numbered metal leg bands were used to help ascertain Fairy Prion, *Pachyptila turtur* (Kuhl, 1820), numbers where several birds used the same rock crevice entrance access, concealing nest sites within. The implications of the 1999 population monitoring results were assessed in relation to total island population estimates for each species resulting from work between 1975 and 1982 (Brothers 1984). Some data of single year counts have been included for comparison with potential future counts.

Searches of all potential petrel-nesting areas were undertaken in each survey period to reveal newly established nesting sites for different species. Extensive spotlight searches together with systematic examination of petrel carcasses were used to further indicate the presence of particular species.

Throughout the seabird monitoring study, that commenced in November 1975, there has been visual evidence of ecological responses to changing climatic conditions. In an attempt to substantiate these observations and identify potential causes of the changes observed, records maintained by the Australian Bureau of Meteorology were examined for trends in climatic conditions around Macquarie Island. Records between 1948 and 2006 were used, but for some weather components, due to problems with data accuracy or missing data in earlier years, records between 1968 and 2006 were examined.

Of particular relevance to the study were changes in the amount and pattern of precipitation at Macquarie Island, specifically in relation to petrel burrow flooding events, and the frequency of events such as sustained lower temperatures, that impact on pest species survival. Weather data were used to ascertain whether the documented trend prior to 1986 of more northerly prevailing wind direction (Adamson *et al.* 1988) has continued, intensified or ceased.

The relationship between rainfall and petrel burrow flooding events is a very important factor in the abundance of petrel species and their relationship to rabbits. Observed flooding events were correlated with rainfall amounts over the previous 24 hours. In addition, eight sets of 3 x 50-cm-deep water table level monitoring holes were dug across representative habitat of varying topography containing burrows. Water levels and associated rainfall amounts could only be obtained intermittently at the Green Gorge study site. To avoid complicating the results, only an equal number of water table readings obtained in each rainfall category have been used. More extensive and complex investigations would be necessary to fully ascertain variability and relationships between water table levels and rainfall. However, where additional duplicate water table readings were included, the pattern of water table response was consistent. This indicates that the small data set available is adequate to demonstrate, at least in a limited way, the relationship between rainfall amounts and burrow flooding events. All water table levels decreased below 50 cm (the depth of each measurement site) were not included for analysis except when determining water table levels relative to a nominal burrow depth of 25 cm. Surface water table levels (0 cm) were included when calculating the average rate of rise or decline in water level.

The limited rainfall observations made at Green Gorge were compared with data from the meteorological station 15 km to the north to ascertain whether water table level fluctuations or observations of burrow flooding events anywhere on the island correlated with rainfall. More precise comparisons between rainfall recorded at the Bureau of Meteorology weather station and Green Gorge were constrained as rainfall records for Green Gorge were taken at slightly different time intervals to those at the station. In addition, rainfall records at Green Gorge were taken to the nearest 1.0 mm with any amount less than 1.0 mm being recorded as trace only, whereas station records were taken to the nearest 0.2 mm. However, this would have created only a minor discrepancy between rainfall observations at the two sites. The Bureau of Meteorology weather station observations are taken near sea level at the island's northern end, and therefore could be atypical of the rest of the island and may not provide data that reflect weather impacts on the island as a whole.

## RESULTS

### White-headed Petrel, *Pterodroma lessonii* (Garnot, 1826)

From 1993–1999, there was an apparent decline in White-headed Petrel burrow occupancy of 20–30% (table 4). Brothers (1984) showed that White-headed Petrels were highly susceptible to disturbance by rabbits, and these figures assume that the seven burrows at Hill 291 in 1999, occupied by both rabbits and petrels, were not retained by the petrel. Rabbit numbers and associated vegetation damage in the vicinity of these monitoring plots remained high relative to other White-headed Petrel nesting areas. Therefore the decline in the number of active burrows in other areas since the 1970s surveys (Brothers 1984) is likely to be due to cat predation, rather than the impact of rabbits. Cat predation of petrels intensified during the period when myxomatosis was introduced to kill rabbits, which resulted in lower food availability for cats in these areas.

**TABLE 1**
**Antarctic Prion monitoring burrow counts on 18 transects at four localities, between 1988 and 1999 on Macquarie Island**

| | Occupied prion burrows | | | Unoccupied prion burrows | | | Prion in rabbit burrows | | | Rabbit only burrows | | | White–headed petrel burrows | | | Burrow occupancy indicated by prion call response | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | | | | Prion burrow | | | Rabbit burrow | | |
| Year | 88 | 93 | 99 | 88 | 93 | 99 | 88 | 93 | 99 | 88 | 93 | 99 | 88 | 93 | 99 | 88 | 93 | 99 | 88 | 93 | 99 |
| ***Prion Lake*** | | | | | | | | | | | | | | | | | | | | | |
| North side | 29 | 51 | 32 | 0 | 0 | 1 | 1 | 0 | 7 | 2 | 2 | 5 | 0 | 0 | 0 | – | 15(1)[1] | 15(3) | – | 0 | 7(3) |
| South side | 27 | 26 | 12 | 0 | 0 | 5 | 1 | 2 | 4 | 0 | 3 | 3 | 0 | 1 | 0 | – | 13 | 3 | – | 2 | 4 |
| East side | 8 | 23 | 6 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 2 | 0 | 0 | 0 | – | 9 | 2 | – | 0 | 1 |
| West side | 22 | 25 | 10 | 0 | 0 | 1 | 0 | 1 | 2 | 0 | 1 | 4 | 0 | 0 | 0 | – | 9 | 8(3) | – | 1 | 2(2) |
| Centre north/ south | 12 | 16 | 10 | 0 | 0 | 0 | 0 | 0 | 2 | 2 | 1 | 5 | 0 | 1 | 0 | – | 3 | 4 | – | 0 | 2(2) |
| Centre east/ west | 13 | 13 | 14 | 0 | 0 | 1 | 1 | 0 | 1 | 0 | 1 | 1 | 0 | 0 | 0 | – | 3 | 8(3) | – | 0 | 1 |
| Total | 111 | 154 | 84 | 0 | 0 | 8 | 4 | 3 | 17 | 4 | 5 | 20 | 0 | 2 | 0 | | 53 | 49 | | 3 | 24 |
| ***Square Lake*** | | | | | | | | | | | | | | | | | | | | | |
| North side | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | – | 0 | | – | 0 | 0 |
| South side | 15 | 9 | 16 | 0 | 0 | 1 | 0 | 4 | 0 | 0 | 5 | 3 | 0 | 0 | 0 | – | 7(2) | 7 | – | 4 | 0 |
| East side | 20 | 8 | 25 | 0 | 0 | 0 | 0 | 3 | 1 | 1 | 12 | 3 | 1 | 0 | 1 | – | 5(2) | 11 | – | 3 | 1 |
| West side | 19 | 14 | 21 | 0 | 0 | 7 | 5 | 4 | 0 | 3 | 21 | 6 | 0 | 1 | 1 | – | 8(1) | 13(2) | – | 4 | 0 |
| Centre north/ south | 16 | 13 | 21 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 11 | 4 | 0 | 2 | 0 | – | 3 | 8(1) | – | 0 | 0 |
| Centre east/ west | 7 | 4 | 9 | 0 | 0 | 0 | 0 | 3 | 1 | 1 | 4 | 1 | 0 | 0 | 0 | – | 5 | 6(2) | – | 3 | 1 |
| Total | 77 | 48 | 92 | 0 | 0 | 9 | 5 | 14 | 2 | 6 | 53 | 17 | 1 | 3 | 2 | | 33 | 50 | | 14 | 2 |
| ***Tulloch Lake*** | 30 | 38 | 56 | 0 | 0 | 13 | 0 | 1 | 12 | 0 | 11 | 29 | 0 | 2 | 2 | – | – | 15(3) | – | 1 | 12(2) |
| Overall Totals | 218 | 240 | 232 | 0 | 0 | 26 | 9 | 18 | 31 | 10 | 69 | 66 | 1 | 7 | 4 | | 86 | 117 | | 18 | 40 |
| ***Prion South*** | – | – | 31 | – | – | 1 | – | – | 6 | – | – | 5 | – | – | 0 | – | – | 21(1) | – | – | 6(2) |

[1] Parentheses indicate additional pairs within same burrow detected by call playback response.

**TABLE 2**
**Comparative rate of Antarctic Prion population change**

| Basis of count | Total counts | | | % change[1] | | |
|---|---|---|---|---|---|---|
| | **1988** | **1993** | **1999** | **1988-93** | **1993-99** | **1988-99** |
| Occupied prion burrows | 218 | 240 | 232 | >10.0 | <3.0 | >6.4 |
| Occupied prion plus prion in rabbit burrows | 227 | 258 | 263 | >13.7 | >1.9 | >15.9 |
| Call playback response from prion burrows | – | 86 | 117 | – | >36.0 | – |
| Call playback response from prion burrows plus rabbit burrows | – | 104 | 157 | – | >51.0 | – |

[1] Calculated using combinations of the burrow count and call playback response data, all sites combined from which data were collected (from table 1).

**TABLE 3**
**Antarctic Prion occupancy rate[1] of rabbit burrows with and without the use of call playback**

| | Occupied prion burrows | Rabbit burrow without prion | Prion occupied rabbit burrows from call playback response | Prion occupied rabbit burrows without call playback response |
|---|---|---|---|---|
| 1988 Prion, Square, Tulloch – transect data combined | 218 | 10 | – | 9 |
| 1993 and 1999 Prion, Square, Tulloch and Prion South transect data combined | 503 | 140 | 66 | – |

[1] Based on selected data from 18 transects at four sites on Macquarie Island between 1988 and 1999 (from table 1).

**TABLE 4**
**Burrow counts at monitoring plots for White-headed Petrels on Macquarie Island in 1993 and 1999**

| Site | Year | Burrow Counts | | | | Total Petrel |
|---|---|---|---|---|---|---|
| | | Occupied | Unoccupied | Petrel and rabbit | Rabbit | |
| Hill 291 | 1993 | 27 | 3 | 0 | 3 | 27 |
| | 1999 | 18 | 0 | 7 | 15 | 25 |
| Sodomy West | 1993 | 22 | 0 | 0 | 0 | 22 |
| 023326 | 1999 | 17 | 0 | 0 | 8 | 17 |
| Sodomy East | 1993 | - | - | - | - | - |
| 025326 | 1999 | 42 | 0 | 0 | 0 | 42 |

Three White-headed Petrel population sites were monitored. Vegetation cover differed between sites over time because of rabbit grazing in two of the three sites. This had a variable influence on petrel abundance. Hill 291 site changed little, if at all, since 1975, as it was almost entirely denuded of tussock. West Sodomy was ungrazed by rabbits from before 1975 until 1993. However, by 1999 the vegetation became thinner and more stunted, possibly due to climatic conditions. Similarly, East Sodomy was dense tussock until at least 1993. However, by 1999 the southern half (with few burrows) had sparse tussock remaining, whereas the northern half was still relatively healthy. If a combination of results from Hill 291 and Sodomy West is indicative of an overall population trend, then from 1993–1999 a decline from 7890 breeding pairs to around 6800 has occurred on the island, assuming the original population estimate (Brothers 1984) remained stable up to 1993. However, the sample size of these two sites is small relative to the total island population.

## Antarctic Prion, *Pachytila desolata* (J.F. Gmelin, 1789)

There appears to have been a 6% increase in the number of occupied burrows between 1988 and 1999. This figure is based on comparison between years, by combining count data of occupied prion burrows from Prion Lake, Square Lake and Tulloch Lake (table 1). Whilst the increase from 1988 to 1993 was 10% and numbers declined by 3% from 1993 to 1999, inter-year comparisons between sites or between individual transects within a site, showed numbers could fluctuate considerably. Much of this fluctuation appeared to be related to rabbit activity, with a decrease in occupied prion burrows occurring when more rabbit burrows were present (table 1). It is not known how many prions utilise rabbit burrows for breeding and given that the rabbit population does fluctuate, prion numbers based on "occupied burrows" may only appear to be fluctuating. If call playback response is indicative of occupancy rate then this can only account for between 7% and 15% being present yet occupying rabbit burrows. This is the proportion of all prion-occupied burrows where prion call response in both 1993 and 1999 was from a rabbit burrow (derived from the counts of occupied prion burrows plus prion in rabbit burrows) (table 1).

If call playback response only is used as the measure of occupancy irrespective of burrow type, then the number of burrows occupied by Antarctic Prions increased by 30% at Prion Lake, 11% at Square Lake or by 18% overall between 1993 and 1999. However, over the same period, based on the sum of "occupied prion burrows" and "prions in rabbit burrows" at Prion Lake, the population decreased by 36%, and increased by 52% at Square Lake (table 1). By using combined area counts the estimate was of an overall 16% increase, 14% occurring between 1988 and 1993 and 2%

between 1993 and 1999 (table 2). This is not evident from an estimate based solely on the call response rate (table 2) Given that the pattern of daily burrow occupancy by individuals that are not actually breeding (deferred or failed breeders, immature or unpaired birds) has been observed to vary greatly in response to daily wind strength and direction as well as moon phase, this is to be expected.

In 1988, the only year when call playback response was not used to differentiate prion-occupied and prion-unoccupied rabbit burrows, the proportion of rabbit burrows assessed as occupied by prions was 47% but this accounted for only 4% of the total count of occupied prion burrows in that year (table 3). Call playback response revealed 32% of rabbit burrows to contain prions, which if added to the total of occupied prion burrows represented 11.6% of the total number of prions counted. The correlation between rabbit burrows assessed as occupied by a prion using manual methods and the voice callback generated from these burrows, was not assessed. Call playback response revealed two or more pairs in occupied prion burrows on 23 out of 180 occasions (15%) but more frequently, 9 in 49 occasions, (22%) in prion-occupied rabbit burrows (table 1). Out of 23 occasions when more than one pair was in the prion burrow, there were two breeding pairs, except for two occasions when three pairs responded. More than one pair was defined by call response from individuals at discernibly different positions within a burrow system.

By the time prion transects were established, rabbits had become very scarce in these areas (Copson 2004). Therefore prion burrow entrances were already becoming increasingly inconspicuous due to vegetation (particularly *Acaena* sp.) becoming taller, denser and more widespread without rabbit grazing, and with increased growth of lichens and mosses at burrow entrances further disguising evidence of occupancy. It is important to note that further habitat changes impacting on monitoring accuracy are likely in the future, especially if rabbits are eradicated.

The absence of burrows on sections of transects or whole transects (e.g., on the north side of Square Lake) was due to unsuitable habitat, usually water-saturated vegetated ground (mires). Few burrows occur in feldmark but this was sparse and only encountered on the Tulloch Lake transect, with the majority of all sites being dominated by a mix of short grassland and herbfields. Some minor effect on future monitoring is likely to result from a recovery of *Pleurophyllum hookeri* Buchanan and, in particular, *Acaena megellanica* (Lam.) Vahl. at these sites. Copson & Whinam (1998) found that, without rabbits, existing species of plants became more luxuriant in the first three years, after which relative species abundance also changed.

Assuming that the 16% average increase in abundance at monitored sites is indicative of overall trends and that the earlier population estimate of 48 900 pairs (Brothers 1984) remained stable up to 1993, the 1999 estimate would be around 56 700 breeding pairs of Antarctic Prions on Macquarie Island.

## Sooty Shearwater, *Puffinus griseus* (J.F. Gmelin, 1789)

A total of 66 colonies of this species were located, with ten of these containing no active burrows in 1999 and one colony with no active burrows on previous counts, again containing occupied burrows (table 5). At 19 sites (identified in table 5 by the number in parentheses following the colony name), burrows occurred in disconnected patches too close to identify separately by grid reference. A total of 40 such patches (discounting the locality identifier patch itself) were identified, many having been created by overall colony deterioration through rabbit activity and vegetation destruction (e.g., at South Coast West)

The incidence of severe rabbit damage was greatest at the time of the 1999 census when 35% (23 of 66 colonies) were affected with a further 18.2% (i.e., 12) being either moderately or lightly damaged. In addition, a further 12 colonies had been subjected to severe rabbit damage at some stage in the past (table 5). Of 18 severely rabbit-damaged colonies, for which comparable counts were made, a 61% decline occurred by 1993, whereas ten severely damaged since 1993 (most of these probably not until 1998 or 1999) had not yet declined. An increase of 13% occurred at the 11 colonies moderately or lightly damaged by rabbits between 1993 and 1999 but a decline of 7% occurred at the five colonies that had not been rabbit-damaged during this period.

Based on total burrow counts the shearwater population has declined by 36% from a pre-1979 estimate of 1806 pairs to a 1999 estimate of 1147 pairs (table 5). However, since 1979 an additional 27 colonies containing 361 pairs have been located, which, if subtracted from the 1999 total suggests that the decline, could be as high as 56.5%. However, at least nine (indicated in table 5) and possibly more of the 27 "new" colonies may contain birds from existing colonies that declined in numbers over this period. There was an 89% decline at one colony alone (South Coast West) that originally contained around one-third of the total island population.

Using only counts taken both prior to 1979 and in 1999 (table 6) at colonies considered most representative and reliable (smaller colony size and lower burrow density enabling better count accuracy), a decline of 20% occurred overall. In this sample, at seven colonies, numbers increased overall by 54% whereas in 22 others an increase of 38% occurred overall. A further sub-sample (indicated in table 6) of only those colonies for which a count was made in each census year indicates a decline in nine of the ten colonies, of 45.8% overall. The same data in table 1 also suggest the majority of the decline occurred in the period between 1979 and 1988, that of 53% over nine of the ten colonies. Subsequently, numbers increased, in seven of the ten colonies from 1988 to 1993 by 33% and by a further 12% in seven colonies from 1993–1999.

## Grey Petrel, *Procellaria cinerea* J.F.Gmelin, 1789

There was no evidence of Grey Petrels attempting to resume breeding on the island until 1993 despite periods of intensive searching since 1976. In November 1993 a total of 11 burrows at four sites (table 7) revealed evidence of having been occupied, feathers of this species being found at each burrow as well as four dead adults that had at least been chewed on, if not killed, by cats. Between 1993 and 1999 the species resumed breeding for the first time and did so successfully in 1999 with signs that four chicks may have fledged from a total of 12 burrows (at seven sites) that revealed signs of occupation (table 7). Based on data in Schulz *et al.* (2005), the chicks could have fledged within several weeks of 23 November 1999 when the first sign of their presence was found.

At two of the three sites occupied in 1993, there was no evidence of the species' presence in 1999. The typical cavernous Grey Petrel burrow at the second colony in table 7 was, in 1999, occupied by a Pacific Black Duck, *Anas superciliosa* J.F. Gmelin, 1789, incubating seven eggs.

All burrows considered to have fledged chicks had abundant evidence of this in the form of obviously well-used nesting sites containing moulted nestling plumage, feather "scales" and, in several instances, egg-shell fragments. With the exception of the duck-occupied site, all other sites were within or adjacent to known colonies of Sooty Shearwaters. This was not surprising considering that the necessary nesting habitat and site characteristics were anticipated to be similar for each species.

There was none of the usual evidence such as dead bird remains left by cats or skuas to indicate the return of this species to the island. The lack of remains of dead birds away from the immediate proximity of burrows, does implicate cats in particular and not skuas as the determinant of this species' fate since skuas preying on White-headed Petrels and Grey Petrels from tussock-covered colonies would take the petrel to open terrain, away from the colony. Also, based on the indications of cat abundance over time (Copson 2002), Grey Petrels may have had little chance to breed successfully until 1997 or 1998.

Grey Petrels have continued breeding in at least some of the 1999 burrows that fledged chicks in 2000 (DPIWE unpublished records) and additional burrows as well as new breeding sites have been subsequently found (Schulz *et al.* 2005).

## Cape Petrel, *Daption capense* (Linnaeus, 1758)

Despite Cape Petrels being seen regularly offshore and occasionally feeding on seal carcasses at the shoreline, the only observations of this species ashore anywhere in the region were small numbers at Anchor Rock nearly 1 km to the northwest of the island's northern end and on Gorilla Rock which was visited regularly by up to six birds in late October 1979. The opportunity to determine the breeding status of this species at these relatively inaccessible localities did not arise. It was immediately adjacent to Gorilla Rock, only several hundred metres off the east side of the island's northern end, that a pair was first reported occupying a suitable nest ledge in mid- to late October 1999 (A. Rooke pers. comm.). However, it was not until 2002 that the first breeding attempt on the island was recorded at this site (Threatened Species Section 2007).

## Soft-plumaged Petrel, *Pterodroma mollis* (Gould, 1844)

By late 1999, despite additional evidence to that reported in Brothers (1984), breeding of Soft-plumaged Petrels had not been confirmed and there were seven more records of cat- or skua-killed adult birds, two of these at locations where birds had previously been found dead (indicated in table 8). At two localities, burrows believed to be those of Soft-plumaged Petrels were found near dead bird remains with all three burrows being empty, although two contained freshly-collected nesting material (i.e., vegetation). These burrows and most remains of dead birds were found in the vicinity of Sooty Shearwater colonies. This added to the difficulty of confirming whether this species was breeding on Macquarie Island. Although further observations have been made since 1999, breeding was not confirmed until the 2006 breeding season (Bryant & Shaw 2007). The records of Soft-plumaged Petrels seen ashore in July at Macquarie Island (Threatened Species Section 2007) adds weight to the argument (Brothers 1984) that such timing of visitation would have made them particularly vulnerable to predators, therefore increasing the likelihood that recent records are indications of yet another petrel species recovering here, rather than newly establishing.

## Blue Petrel, *Halobaena caerulea* (J.F. Gmelin, 1789)

Prior to 1999, all searches of colonies for Blue Petrels, including that of Brothers (1984), indicated that there was little or no prospect of a successful outcome from breeding attempts. For example, at the Hurd Point colony in November 1988 only one incubating adult was found aside from three fresh-killed, partly-eaten birds. In November 1993 out of 31 burrows, only four contained an adult incubating, two had a pair of birds without an egg and the remainder were empty, four of these with chewed eggs at entrances. There was also extensive damage to tussock and *Cotula plumosa* Hook. f. by rats. At Douglas Point in November 1988, incubating birds were readily located despite abundant evidence of rats. However, by early December 1993 there were partly-eaten eggs and chick carcasses on the surface of the colony and only three birds brooding new chicks in the 15 nests that it was possible to check. At Green Gorge colony in late November 1993, nest sites were unoccupied, and there was no sign of rats, which is usual only at this colony. Therefore the four abandoned eggs found here had remained uneaten.

In 1988 the Langdon Point Blue Petrel colony did not have the usual skua territory adjacent to it and remains of only five petrels were found here whereas in 1993 there were the remains of 34 petrels. The colony was not visited in 1993 due to rough seas. In 1988 it was typically rat-damaged, but in 1999 there was no evidence of rats being present. As a consequence, on 26 January 2000, for the first time, there were indications that Blue Petrel chicks were being reared successfully. One burrow contained a full-grown chick with only a small amount of down remaining at the nape of its neck. There were also seven burrows, excavated by skuas, that had contained similarly advanced chicks. However, at other breeding colonies when visited between early December and late January there were abundant signs of rats but no recent petrel activity.

To explain why the Langdon Point colony had no rats in 1999, meteorological data in preceding years were examined. These revealed a prolonged, severe cold period in 1995, the severity of which had not occurred since records began in April 1948. In late July 1995 there was the most persistently cold week ever recorded with nine consecutive days averaging below 0°C. This month was also the coldest calendar month and had the coldest day (-4.8°C) on record. The combination of low Blue Petrel abundance in July (Brothers 1984) and the dominant plant species being scarce here for both rat protection and food would have increased the vulnerability of rats present. While rats may have been eliminated from all other Blue Petrel colonies by the same event, the Langdon Point colony was the only site without rats four breeding seasons later, probably due to the exposed rock and expanses of water which rats need to negotiate to repopulate the area. Green Gorge colony, situated on a rock

## TABLE 5
**Locality details, burrow counts and habitat condition for Sooty Shearwater colonies on Macquarie Island between 1975 and 1999**

| Colony location | Locality | Year 1975-1979 | | | | Year 1988 | | | | Year 1993 | | | | Year 1999 | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Burrow | Rabbit damage | | | Burrow | Rabbit damage | | | Burrow | Rabbit damage | | | Burrow | Rabbit damage | | |
| | grid ref | count | none | past | current activity | count | none | since 1979 | current activity | count | none | since 1988 | current activity | count | none | since 1993 | current activity |
| North Head | 116617 | 20 | √ | | | | | | | 0 | √ | | | 0 | √ | | |
| North Head | 113617 | | | | | | | | | 10 | √ | | | 6 | √ | | |
| North Head | 116619 | 20 | √ | | | | | | | 0 | √ | | | 0 | √ | | |
| North Head | 114619 | 10 | √ | | | 0 | | | Severe | 0 | √ | | | 0 | √ | | |
| North Head | 113625 | 10 | √ | | | 0 | | | Severe | 0 | √ | | | 0 | √ | | |
| North Head | 113626 | 10 | √ | | | 13 | | | Mod | 16 | √ | | | 14 | √ | | |
| North Head | 114627 | 10 | √ | | | 4 | | | Mod | 10 | √ | | | 8 | √ | | |
| North Head | 115624 | 10 | √ | | | 0 | | | Severe | 0 | √ | | | 0 | √ | | |
| North Head | 116623 | 20 | √ | | | 3 | | | Severe | 4 | √ | | | 0 | √ | | |
| North Head | 116620 | 20 | √ | | | 14 | | | Mod | 18 | √ | | | 0 | √ | | |
| Perseverance Bluff | 101604 | | | | | | | | | | | | | 2[2] | | | Light |
| Gadget Gully | 107602 | 23 | √ | | | 9 | | | Light | 10 | √ | | | 9 | √ | | Mod |
| Halfway Hill (4)[1] | 109596 | | | | | | | | | 36 | √ | | | 30 | | | Severe |
| Halfway Hill South | 109593 | | | | | | | | | 11 | √ | | | 19 | | | Severe |
| Sandy Bay North | 102558 | | | | | | | | | | | | | 7[2] | | | Light |
| Sandy Hut (3) | 098546 | 49 | | | Severe | 8 | | | Mod | 17 | √ | | | 19 | | | Severe |
| Sandy Bay South | 097543 | | | | | | | | | | | | | 8[2] | | | Light |
| Brothers Point (8) | 100525 | 20 | √ | | | 4 | √ | | | | √ | | | 20 | √ | | |
| Brothers Point | 100526 | 40 | √ | | | 36 | √ | | | | √ | | | 19 | √ | | |
| Brothers Point | 100527 | 30 | √ | | | 9 | √ | | | | √ | | | 15 | √ | | |
| Brothers Point Trig | 103523 | | | | | 22 | √ | | | 18 | √ | | | 17 | | | Light |
| Brothers Point –all 8 sites | | 90 | | | | | | | | 78 | | | | 82 | | | |
| Green Gorge | 087462 | 36 | √ | | | 18 | | | Light | 29 | √ | | | 35 | | | Mod |
| Green Gorge South (2) | 085452 | | | | | | | | | 7 | √ | | | 11 | | | Severe |
| Saddle Point (2) | 076426 | 5 | | | Severe | | | | | 2 | √ | | | 0 | | | Severe |
| Saddle Point | 076427 | 19 | √ | | | | | | | | √ | | | 8 | | | Severe |
| Saddle Point | 078428 | 20 | √ | | | | | | | | √ | | | 18 | | | Severe |
| Saddle Point | 078444 | 7 | √ | | | | | | | 7 | √ | | | 11 | | | Severe |
| Saddle Point | 086457 | 13 | √ | | | | | | | | √ | | | 0 | | | Severe |
| Lusitania North | 059389 | | | | | | | | | | | | | 4[2] | √ | | |
| Lusitania North | 058387 | | | | | | | | | | | | | 8[2] | √ | | |
| Lusitania North | 053370 | | | | | | | | | | | | | 14 | √ | | Light |

| | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Lusitania Hut | 054366 | 50 | √ | | | 13 | | | Severe | 44 | √ | | | 38 | √ | | |
| Mt Jeffries | 055355 | | | | | | | | | 1 | | | Light | 0 | √ | | |
| Mt Jeffries | 055346 | | | | | | | | | 13 | | | Light | 18 | √ | | |
| Mt Jeffries | 054346 | | | | | | | | | 7 | | | Light | 6 | √ | | |
| Mt Jeffries | 054345 | | | | | | | | | 3 | | | Light | 5 | √ | | |
| Mt Jeffries | 050336 | | | | | | | | | | | | | 13[2] | | | Severe |
| Hurd Scree East (3) | 043314 | 20 | | | Severe | | | | | | √ | | | 36 | | | Severe |
| Hurd Scree West | 036311 | 10 | √ | | | | | | | 19 | | | Severe | 11 | | | Severe |
| South Coast East | 027312 | 60 | | | Severe | 22 | | | Severe | 47 | √ | | | 18 | | | Severe |
| South Coast West | 024312 | 500 | | | Little | 100 | | | Severe | 79 | | | Severe | 53 | | | Severe |
| S. W. Point | 012317 | 24 | √ | | | 26 | √ | | | 16 | √ | | | 12 | √ | | |
| Petrel Peak | 013319 | | | | | | | | | | | | | 7[2] | √ | | |
| Sodomy West (3) | 020328 | 100 | √ | | | | | | | 14 | √ | | | 39 | | | Severe |
| Cape Star (2) | 017341 | 100 | √ | | | 35 | | | Severe | 27 | √ | | | 30 | | | Severe |
| Precarious Point (3) | 024356 | 48 | √ | | | | | | | 41 | | | Severe | 88 | √ | | |
| Precarious North | 028367 | | | | | | | | | | | | | 1 | | | Severe |
| Rockhopper Point | 026395 | 10 | √ | | | | | | | 14 | √ | | | 39 | | | Severe |
| Rockhopper North | 026396 | | | | | | | | | | | | | 33 | √ | | |
| Cape Toutcher (3) | 029406 | | | | | | | | | | | | | 46 | √ | | |
| Sandell | 044419 | | | | | | | | | | | | | 7[2] | | | Light |
| Mt Waite West (2) | 041458 | | | | | | | | | 30 | | Mod | | 22 | | | Mod |
| Selleck Bay North (4) | 041472 | 25 | √ | | | | | | | 30 | √ | | | 34 | | Severe | |
| Skua Lake West (2) | 041479 | 23 | √ | | | | | | | 11 | | | Severe | 17 | | Mod | |
| South Soucek Bay (3) | 042478 | | | | | | | | | | | | | 29 | | | Severe |
| Soucek Bay | 047486 | | | | | | | | | | | | | 4 | | | Severe |
| Aurora Point Lower | 047496 | 100 | √ | | | | | | | 6 | | | Severe | 6 | | Mod | |
| Aurora Point Upper (3) | 047495 | 3 | | Severe | | 2 | | | Light | 0 | | | Severe | 1 | √ | | |
| Cormorant Point (3) | 066529 | | | | | 4 | √ | | | 6 | √ | | | 19 | | | Severe |
| Flat Creek North | 065532 | | | | | | | | | | | | | 4[2] | √ | | |
| Mawson Point (2) | 065543 | 20 | √ | | | 12 | √ | | | 17 | √ | | | 18 | | | Light |
| Bauer North (2) | 074564 | | | | | | | | | 5 | √ | | | 21 | | | Severe |
| Langdon West | 065578 | 200 | √ | | | | | | | 28 | √ | | | 26 | | | Severe |
| Langdon East | 070576 | 20[2] | √ | | | 4 | | Severe | | 6 | √ | | | 14 | | | Mod |
| Handspike Corner (6) | 083612 | 91 | √ | | | 68 | | Severe | | 93 | √ | | | 113 | | | Light |
| TOTALS | | 1806 | | | | 426 | | | | 830 | | | | 1147 | | | |

[1] Parentheses indicate number of separate burrow patches
[2] Newly established colony.

**TABLE 6**
**Sooty Shearwater population trends[1] on Macquarie Island prior to 1979 and in 1999**

| Colony | Grid | 1979 | 1999 | Count Difference | |
|---|---|---|---|---|---|
| | | | | Up | Down |
| S W Point[2] | 012317 | 24 | 12 | | 12 |
| Handspike Corner[2] | 083612 | 91 | 113 | 22 | |
| Precarious Point | 024356 | 48 | 88 | 40 | |
| Gadget Gully[2] | 107602 | 23 | 9 | | 14 |
| Sellick Bay North | 041472 | 25 | 34 | 9 | |
| Skua Lake West | 041479 | 23 | 17 | | 6 |
| Aurora Pt upper[2] | 047495 | 3 | 1 | | 2 |
| Cape Star[2] | 017341 | 100 | 30 | | 70 |
| Mawson Point[2] | 065543 | 20 | 18 | | 2 |
| Brothers Point | 100524 | 90 | 82 | | 8 |
| Langdon East[2] | 070576 | 20 | 14 | | 6 |
| North Head | 116620 | 20 | 0 | | 20 |
| | 116623 | 20 | 0 | | 20 |
| | 115624 | 10 | 0 | | 10 |
| | 114627 | 10 | 8 | | 2 |
| | 113626 | 10 | 14 | 4 | |
| | 113625 | 10 | 0 | | 10 |
| | 114619 | 10 | 8 | | 2 |
| Green Gorge[2] | 088462 | 36 | 35 | | 1 |
| Lusitania Hut[2] | 054366 | 50 | 38 | | 12 |
| Sandy Hut[2] | 098543 | 49 | 19 | | 30 |
| Hurd Scree West | 036311 | 10 | 11 | 1 | |
| Sodomy West | 020328 | 100 | 39 | | 61 |
| Rockhopper Point | 026395 | 10 | 39 | 29 | |
| Saddle Point | 076426 | 5 | 0 | | 5 |
| | 076427 | 19 | 8 | | 11 |
| | 076428 | 20 | 18 | | 2 |
| | 078444 | 7 | 11 | 4 | |
| Hurd Point East | 043314 | 20 | 36 | | 16 |
| Total | | 883 | 702 | 109 | 259 |

[1] Based on colonies for which comparable counts were made prior to 1979 and in 1999 (from table 5).
[2] Colonies with high count reliability for all count years.

**TABLE 7**
**Number of burrows, breeding evidence and site locality of Grey Petrels at Macquarie Island in 1993 and 1999**

| Site locality | Burrows with occupancy evidence | | Burrows from which chicks had fledged | |
|---|---|---|---|---|
| | 1993 | 1999 | 1993 | 1999 |
| Green Gorge 087462 | 8 | 2 | 0 | 1 |
| Green Gorge 088463 | 0 | 1 | 0 | 0 |
| Green Gorge Sth 085452 | 1 | 0 | 0 | 0 |
| Saddle Point 079434 | 1 | 0 | 0 | – |
| Brothers Point 103523 | 1 | 2 | 0 | 0 |
| Brothers Point 100524 | 0 | 1 | 0 | 0 |
| North Head 114618 | 0 | 3 | 0 | 2 |
| North Head 115627 | 0 | 2 | 0 | 1 |
| North Head 114627 | 0 | 1 | 0 | 0 |

**TABLE 8**
**Observation details and locality of Soft-plumaged Petrel records on Macquarie Island between 1982 and 1999**

| Date | Location | Dead Adults | Burrows |
|---|---|---|---|
| 03/11/93 | Saddle Point 079428[1] | 1 | |
| 05/11/93 | Hill 291 escarpment 033311 | 1 | 1 |
| 06/11/93 | South Coast 027312 | 1 | 2 |
| 28/11/93 | Brothers Point 103526 | 1 | |
| 01/12/93 | Green Gorge 088462[1] | 1 | |
| 02/12/93 | Green Gorge 087465 | 1 | |
| 16/01/00 | Sandell Bay 045422 | 1 | |

[1] Localities at which evidence of presence had also been found prior to 1982.

stack, although less distance offshore than the Langdon Point colony, provides another example of physical obstacles to rats. At Green Gorge colony the combination of the open pebble beach and water barrier is the probable explanation for this colony always having been rat-free. The rat-free status of the Green Gorge offshore colony probably explains why Blue Petrels' first attempt to recolonise mainland Macquarie Island was at this location. This new onshore colony at Green Gorge, containing approximately 20 burrows, was found in November 1999 on a low elevation spur, lightly covered with tussock and *Stilbocarpa polaris* (Hombr. & Jacq.)A.Gray, with another smaller patch of burrows on a nearby slope. Although several eaten eggs and dead adults were present, one adult incubating an egg was also found out of ten burrows checked.

Weather-induced rat-free periods on offshore stacks may explain how Blue Petrels have survived despite little or no evidence of successful breeding in the past. Previously, an hypothesis for how this species managed to persist despite no evidence of successful breeding, was that numbers were being replaced from an unidentified colony on Bishop and Clerk islets, 33 km south of Macquarie Island (Brothers 1984). However, the survey there in 1993 (Brothers & Ledingham 2008) did not locate this species. One colony known to exist that has not been visited is on a relatively large offshore stack in Caroline Bay (grid reference 017327) and is likely to be consistently rat-free. It may contain several hundred birds. Several Blue Petrels sighted over slopes at Macquarie Island's northern end in late November 1999, at 2230 hours, in an area where they had previously not been seen, indicate increased or more widespread activity by this species. A dead adult was found within a burrow in mid-January 2000 at South Double Point (grid reference 035444) amongst abundant sign of rats. Previously, the only evidence here of activity had been the cushion-forming plant *Colobanthus muscoides* Hook.f. that appeared to have once supported a network of interconnected burrows.

### Fairy Prion, *Pachyptila turtur* (Kuhl, 1820)

Fairy Prion population recovery in response to vertebrate pest management was indicated in late November 1999 by an estimated 10–20 pairs recorded occupying rock and cliff caverns at the island's northernmost tip. Because the sites were inaccessible, breeding could not be confirmed, although a large chick was subsequently seen at this location in 2001 (Threatened Species Section 2007). Also, on 12 December 1999, during call playback at Antarctic Prion burrows on the Prion Lake monitoring site, a Fairy Prion responded. The breeding status of this particular bird was not ascertained at the time, as this would have entailed unwarranted burrow excavation. These were the first and only indications of this species reoccupying Macquarie itself, although very small numbers have continued breeding since on offshore stacks (Brothers 1984) as well as several pairs breeding on Bishop and Clerk islets (Brothers & Ledingham 2008). On Bishop and Clerk islets, with very scarce suitable habitat and strong inter-specific competition for alternative habitat, little opportunity exists for any increase in this population. On 25 November 1988, three eggs were being incubated at the offshore Langdon Point colony and it was here, on 26 January 2000 that one large down-covered chick was recorded. This is the first record of a Fairy Prion egg having hatched, and coincided with there being no rats at this locality for the first time.

### South Georgian Diving-Petrel, *Pelecanoides georgicus* Murphy & Harper, 1916

The only evidence of this species breeding on Macquarie Island itself was a fully-fledged chick amongst tussock at the southern end of the island on 23 December 1999. Apart from this record, only three burrows on Bishop and Clerk islets are known of in the region (Brothers & Ledingham 2008).

### Common Diving-Petrel, *Pelecanoides urinatrix* (J.F. Gmelin, 1789)

The burrows occupied by Common Diving-Petrels (Brothers 1984) were searched in the 1988 and 1999 breeding seasons but no birds were located. Burrow habitat was very fragile which limited the extent of searching, so it is inconclusive that none was present. The only record for the region of a hatched egg was in the one burrow that is on Bishop and Clerk islets (Brothers & Ledingham 2008).

### Wilson's Storm-Petrel, *Oceanites oceanicus* (Kuhl, 1820)

One burrow, possibly belonging to Wilson's Storm-Petrel, based on similarities with the three Wilson's Storm-Petrel burrows and surrounding habitat on Bishop and Clerk islets (Brothers & Ledingham 2008) was found on 25 January 2000. It was beneath *C. muscoides* on a very small offshore stack (Grid reference 101546) off Macquarie Island's east coast. The nest chamber contained one Black Rat, which,

considering the limited area of the stack, can only have been a brief visitor there. No burrows were found at this locality in previous years.

## Vagrant Petrels

Apart from the previously mentioned petrel species there were only three other species encountered in the Macquarie Island region. The Southern Fulmar, *Fulmarus glacialoides* (A. Smith, 1840), Antarctic Petrel, *Thalassoica antarctica* ( J.F. Gmelin, 1789) and Short-tailed Shearwater, *Puffinus tenuirostris* (Temminck, 1835), were all regularly encountered but only ever as beach-cast remains. When other petrel remains have been found it has been indicative of the species breeding on the island but this is not the case for these three. Likewise, only Snow Petrels, *Pagodroma nivea* (G. Forster, 1777), remains have been found (Scofield & Wiltshire, 2004).

## Macquarie Shag *Leucocarbo atriceps purpurascens* Brandt, 1837

Although not substantiated by diet studies of cats (Jones 1977) or Weka (Brothers & Skira 1984), circumstantial evidence of regular cat and Weka activity in the vicinity of accessible Macquarie Shag colonies suggests that Macquarie Shags may be a food source for these introduced vertebrates. Such a locality and time-limited food source is not so readily detected with diet studies, unlike more widespread and constantly available prey. Macquarie Shag eggs and small chicks would have been relatively easy prey not only for Weka but for rats, and cats could possibly take small and large chicks but not adult shags.

Macquarie Shag colony count data (table 9) indicate that the number of breeding pairs fluctuates annually. The 1999 total of breeding pairs at predator-restricted access sites was similar (174) to the highest total in previous years of 167 based on all sites for which a count was made in each year. The same comparison for all open access colonies provides no clear evidence of a trend. Macquarie Shag populations are generally regulated by inundation of nests by high seas as well as by foraging restrictions caused by the islands' bathymetry and strong winds (Brothers 1985). The combined impact of these influences makes it impossible to accurately determine the impact that introduced predators may have had.

## Kelp Gull, *Larus dominicanus* M.H.K. Lichtenstein, 1823

The most significant evidence of a reduction in introduced predator impact on Kelp Gulls was the first record of a nest containing two eggs found on a steep slope in tall tussock grassland habitat 50 m above sea level on 24 November 1999. Previously, all nests located were situated on elevated shoreline or offshore rocks where nests were either secure from, or more easily defended against, Weka, cats and also Great Skuas *Stercorarius antarctica lonnbergi* (Matthews, 1912). However, in these localities breeding attempts were often unsuccessful because of inundation by high seas. Also, although shoreline or offshore rock sites were more secure from Weka and cats, the opportunities for skua predation on gull eggs or chicks were greater at these sites than at nests with protective vegetation cover nearby. Considering the lack of access by cats to Kelp Gulls throughout the monitoring period, it is understandable that Kelp Gulls were seldom indicated as prey in the gut contents of cats (0.2%, or two occasions in 966 samples DPIWE unpublished data). It may be relevant that, although Weka had been eradicated for ten years, it was not until all but the last few cats had been eliminated, that gulls were known to have made an attempt at nesting in a high predator-risk location.

Counts indicate that there has been little or no change to the distribution and abundance of this species over time. The highest count ever of birds in first-year plumage was made on 25 January 2000 when 35 were seen together in a flock with 26 birds in second-year to pre-adult plumage. While this flock may not have contained every bird on the island from these age classes, it is considered indicative of the small population and low fledgling success of Kelp Gulls on Macquarie Island.

In 1999, a comprehensive whole island census recorded 135 nests with similar numbers on east (69 pairs) and west (66 pairs) coasts. At every site there was only one nest but this was often defended by a number of Kelp Gulls against intrusion. It is possible that, such nesting behaviour became a necessity in response to Weka and cats and, as a result, the number of gulls declined as a consequence of lower breeding success. It remains to be seen whether Kelp Gulls will respond to cat and Weka eradication by nesting in greater abundance in larger colonies or simply continue using widely scattered locations in the traditionally more secure sites offshore, both ultimately leading to an increase in abundance of this species.

## Antarctic Tern, *Sterna vittata bethunei* Buller, 1896

Records of observations made during visits to the island between 1975 and 2000 indicated that Antarctic Terns breed annually at regular sites, in very low and variable numbers, with most breeding attempts failing to produce a fledgling. Eggs are laid from early November to mid-January at widely scattered localities. Antarctic Terns nest on inaccessible offshore stacks with a proportion of non-breeding birds also in attendance, both factors making reliable counts difficult. Breeding attempts were noted to be often unsuccessful in early incubation, possibly due to Great Skua and Kelp Gull predation, and fledglings were noted to be present in a colony simultaneously with as yet unhatched eggs. In general though, breeding was often unsuccessful due to nesting sites being vulnerable to high seas.

Pairs often nested in isolation, although up to five nests were encountered at a site. The very limited, apparently preferred feeding habitat in relatively calm, shallow waters such as inshore reef-protected areas of Sandell Bay and in the region between Langdon Bay and Eagle Bay does not exist anywhere else around the island. It is adjacent to these feeding localities that groups of nesting birds occur. However, along the islands' comparatively sheltered east coast, there are fewer and smaller similar feeding environments, and only isolated pairs occur there. Of 11 known breeding sites six are on the west coast, five on the east coast, and the highest count in any year was in 1999 with 19 pairs nesting. No more than four fledglings have been seen in any year. Terns, like Kelp Gulls, are likely to have been greatly affected first directly and then indirectly by introduced predators. However, they may benefit little from the eradication of cats and Weka because rats continue to pose a threat sufficient to discourage terns from nesting at sites more secure from destruction by high seas (Schulz & Gales 2004). Even nesting sites just offshore, with

inadequate habitat for rats, are visited by rats. Therefore, in a number of nesting locations, if high seas do not kill chicks or eggs, rats do. At three sites vulnerable to high seas no fledglings were produced in any of the six breeding seasons of records between 1975 and 1999 (although in two of these seasons no breeding attempt occurred or was so brief as to go undetected).

A 2003 census of 24 pairs (Schulz & Gales 2004) appears not to be related to the number of nests present and, considering the incomplete nature of all counts, it is difficult to identify any population trend. It is likely that the suspected limited foraging habitat will limit population increase therefore making it difficult to be able to draw conclusions about any relationship between tern populations and vertebrate pest management.

## Great Skua, *Stercorarius antarctica lonnbergi* (Matthews, 1912)

Rabbits have had a major influence on the distribution (Skira 1984) and probably also abundance of Great Skuas even though limited specific data were obtained for this species between 1975 and 1999. Skuas seem to be the only indigenous species that has benefited from any of the vertebrate pests. Prior to the introduction of Myxoma virus to control rabbits in 1978 a significant portion of skua breeding coincided with areas of high rabbit abundance, particularly on the plateau, because rabbits (especially young ones) were a major prey item of skuas (Jones & Skira 1979). The correlation of greater skua abundance with a large rabbit population was supported by the considerable decline in the incidence of skuas rearing two chicks and decrease in overall skua breeding success when rabbit numbers were lowest, as observed in 1988 (Copson 2004).

Skua dependence on rabbits as a food source could have resulted in reduced predation on petrels. However, it was because of rabbits that skuas became more widely distributed throughout petrel nesting localities, and this in turn exposed more petrels to possible skua predation. Following the introduction of Myxoma virus, skua nesting became more closely aligned with food sources such as penguin colonies, and there was a noticeable decline in skua-killed petrel remains. It is possible that the abundance of rabbit kittens, hunted by skuas mostly in daytime, may have made the largely-nocturnal activity of petrel hunting unnecessary. In 1999, skuas had a greater impact than in other breeding seasons on certain indigenous species (e.g., one Southern Giant-Petrel, *Macronectes giganteus* (J.F. Gmelin, 1789) colony alone, had 33 eggs taken by skuas from 80 nests). As a result of pest management actions, there may be readjustments between indigenous species that do not necessarily result in greater abundance for each of those species.

If skuas, particularly their chicks, had been vulnerable, as one would expect, to cat predation, then the skua population would have been decimated. Jones (1977) indicates that skuas are not vulnerable to cat predation, and even during the time interval when there was least availability of rabbits as prey for cats, there was no evidence from examination of cat diet studies that skuas were eaten (G. Copson pers. comm.) It is likely that skuas will have increased predator impact above natural rates of predation on indigenous species, during periods of readjustment in skua breeding distribution and abundance as a response to changes in rabbit abundance. Therefore, keeping rabbit numbers as low as possible, prior to an eradication attempt is important. The serious impact on petrels as a consequence of petrel colony habitat destruction from rabbit grazing is far more detrimental to petrels than any increased skua predation predicted to occur during the process of rabbit eradication.

## Impacts of changing rainfall patterns on burrow-nesting petrels

Since 1988, there has been an impression of brief but heavier rainfall events in comparison with previous constant, but more continuous, light precipitation. The following observations are believed to be a result of variations in the daily rainfall pattern:

- from 1988 onward, petrel colonies in tussock habitat were becoming increasingly conspicuous, as surrounding vegetation attained a drier, sparser appearance contrasting with the usual lush, green petrel colony vegetation. In recent years this trend has given the island a more yellow-brown, rather than yellow-green overall appearance;
- by 1999, gully erosion of walking track sections had become apparent and, rather than being associated with any increase in track usage (Dixon 2001), appeared to result from heavier rainfall occurring in shorter periods of time;
- in 1999, mud-beds in certain temporary lakes or tarns were observed undergoing periods of drying-induced cracking for the first time; and
- by 1993, and especially so by 1999, rabbits appeared to be favouring steep and therefore better-drained coastal slopes. There appeared to be a greater frequency of burrow flooding in other sites.

In observations of seven petrel burrow flooding events the least amount of rainfall to cause flooding was 6.4 mm over the preceding 24-hour period. The relationship between burrow flooding and the rainfall that occurred prior to the 24-hour period immediately before the flooding event was not determined. However, the relationship between a specific amount of precipitation and burrow flooding may be complicated by a prolonged dry or wet interval occurring more than 24 hours before burrow flooding (table 10). This may mean that the average of 13.4 mm of rain in the 24 hours preceding the seven flood events reported here is indicative of the amount of rainfall required to flood burrows. Water table levels in response to rainfall (tables 10,11,12) also indicate that when approximately 13 mm of rainfall falls within a 24-hour period, water table rise was sufficient to inundate burrows (table 12). However, when only 5 mm of rain fell, burrows were not inundated and the water table fell. Unfortunately no data were obtained for water table response to rainfall between 5 mm and 13 mm in a 24-hour period to ascertain a more accurate relationship. In addition, because burrow flooding may have commenced prior to monitoring, the amount of rainfall responsible for inundation of burrows is probably less than the average of 13.4 mm derived from weather station records.

The seven burrow flooding events were correlated with the rainfall records at the station. However, there can be significant differences between rainfall at different localities on the island (table 13), such as with increasing altitude where weather conditions can be cooler, windier and wetter (Tweedie 2000). Over a 244-day period, there was a 16% difference in total rainfall recorded at Green Gorge (507.5 mm) and the station (601.2 mm). However, over shorter time intervals there can be more significant differences in

**TABLE 9**
**The number of Macquarie Shag breeding pairs at Macquarie Island between 1979 and 1999**

| | Census Year | | | |
|---|---|---|---|---|
| | 1979 | 1988 | 1993 | 1999 |
| *Open access to introduced predators* | | | | |
| Handspike Point | 320 | 343 | 364 | 417 |
| Red River | 18 | 44 | 0 | 56 |
| Saddle Point | 35 | 31 | 18 | 26 |
| Lusi South | 16 | 18 | 19 | 18 |
| Green Point | 92 | 128 | 89 | 132 |
| Open access total | 481 | 564 | 490 | 649 |
| *Limited access to introduced predators* | | | | |
| Magnetic Rock | 27 | 37 | 5 | 33 |
| Green Gorge | 16 | 0 | 0 | 0 |
| Hurd Point (1) | 3 | 0 | 0 | 0 |
| Hurd Point (2) | 3 | 0 | 0 | 0 |
| Hurd Point (3) | 7 | 0 | 0 | 0 |
| Sandell Bay | 9 | 4 | 0 | 0 |
| Davis Point | 14 | 0 | 12 | 17 |
| Mt Waite[1] | 7 | – | 24 | 29 |
| Cormorant Point | 20 | 34 | 27 | 46 |
| Precarious Point | 0 | 0 | 0 | 34 |
| Mawson Point | 25 | 24 | 16 | 18 |
| Bauer Bay | 18 | 0 | 0 | 0 |
| Langdon Point | 25 | 0 | 31 | 26 |
| Cape Toutcher[1] | – | – | – | 26 |
| Limited access total | 174 | 99 | 115 | 229 |
| Overall total | 660 | 663 | 605 | 878 |

[1] Dash indicates colony not visited. Bishop and Clerk islets colony excluded.

**TABLE 10**
**Changes in water table level[1] in typical prion and rabbit burrowing habitat on Macquarie Island**

| | | Rainfall (mm) in preceding 24-hr period | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | 0[2] | | <1 | | 5 | | 13 | |
| Site | Hole | A[3] | B | A | B | A | B | A | B |
| 1 | 1 | 37[4] | 50+ | 50+ | 50+ | 31 | 50+ | 50+ | 36 |
| | 2 | 50 | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ | 50 |
| | 3 | 0 | 0 | 1 | 4 | 0 | 0 | 3 | 0 |
| 2 | 1 | 39 | 36 | 22 | 24 | 15 | 19 | 33 | 27 |
| | 2 | 24 | 15 | 21 | 25 | 0 | 11 | 11 | 11 |
| | 3 | 41 | 44 | 38 | 42 | 38 | 40 | 50 | 50 |
| 3 | 1 | 0 | 7 | 6 | 7 | 0 | 7 | 9 | 0 |
| | 2 | 0 | 8 | 8 | 14 | 0 | 4 | 15 | 0 |
| | 3 | 6 | 12 | 15 | 19 | 0 | 10 | 19 | 9 |
| 4 | 1 | 3 | 20 | 19 | 20 | 0 | 10 | 25 | 15 |
| | 2 | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ |
| | 3 | 50+ | 50+ | 50+ | 50+ | 41 | 50+ | 50+ | 50+ |
| 5 | 1 | 16 | 24 | 30 | 30 | 9 | 25 | 27 | 11 |
| | 2 | 45 | 33 | 34 | 39 | 19 | 28 | 36 | 36 |
| | 3 | 50 | 50+ | 47 | 48 | 28 | 50+ | 50+ | 50+ |
| 6 | 1 | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ |
| | 2 | 50+ | 50+ | 50+ | 50+ | 30 | 50+ | 50+ | 40 |
| | 3 | 50+ | 50+ | 50+ | 50+ | 47 | 50+ | 50+ | 44 |
| 7 | 1 | 36 | 50 | 50+ | 29 | 26 | 50+ | 50+ | 31 |
| | 2 | 15 | 24 | 25 | 50+ | 0 | 24 | 27 | 16 |
| | 3 | 50 | 50+ | 50+ | 50+ | 30 | 35 | 50+ | 43 |
| 8 | 1 | 9 | 24 | 24 | 24 | 7 | 22 | 28 | 15 |
| | 2 | 50 | 38 | 42 | 50+ | 16 | 32 | 50+ | 50+ |
| | 3 | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ | 50+ |

[1] In response to varying amounts of rainfall in the preceding 24-hour period.
[2] Rainfall categories: <1 mm – insufficient to cause burrow flooding, 5 mm – possible flooding, 13 mm or more – flooding likely.
[3] Water levels at: A – start observation period; B – end 12-hr observation period.
[4] 0 cm – surface, 50 cm – maximum depth reading. Water levels below this depth are shown as 50+.

**TABLE 11**
**Water table fluctuations**

| Water table level change | | Amount of rainfall (mm) in preceding 24 hr[1] | | | |
|---|---|---|---|---|---|
| | | 0 | <1 | 5 | 13 |
| Increase | number of records | 4 | 0 | 0 | 9 |
| | average amount (cm) | 9 | – | – | 10.3 |
| | range (cm) | 3–12 | – | – | 3–16 |
| Decrease | number of records | 9 | 10 | 12 | 0 |
| | average amount (cm) | 9.7 | 3.1 | 9.8 | – |
| | range (cm) | 3–17 | 1–6 | 2–24 | – |
| Stable | number of records | 1 | 2 | 1 | 3 |
| Indeterminate | number of records | 10 | 12 | 11 | 12 |
| | all records | 24 | 24 | 24 | 24 |

[1] Derived from table 10.

**TABLE 12**
**Water table level[1] response to rainfall over a 24-hour period**

| Number of occasions of water table level status | Rainfall (mm) | | | |
|---|---|---|---|---|
| | 0 | <1 | 5 | 13 |
| remained above 15 cm only | 4 | 3 | 6 | 4 |
| remained above 25 cm only | 9 | 8 | 10 | 6 |
| remained below 15 cm only | 16 | 20 | 14 | 16 |
| remained below 25 cm only | 15 | 15 | 11 | 15 |
| increased to above 15 cm only | 1 | 0 | 0 | 4 |
| increased to above 25 cm | 0 | 0 | 0 | 3 |
| decreased to below 15 cm | 3 | 1 | 4 | 0 |
| decreased to below 25 cm | 0 | 1 | 3 | 0 |

[1] In relation to a depth at which smaller petrel burrows may be flooded (15 cm) and to a depth at which rabbit and larger petrel burrows may be flooded (25 cm) based on water level measurements of three holes at eight sites (from table 10).

**TABLE 13**
**Rainfall observation comparisons between 15 km distant localities on Macquarie Island over a total of 244 days in 1976**

| Observation period (day/mth) | Green Gorge rainfall (mm) | Station rainfall (mm) |
|---|---|---|
| 10/03–11/03 | 5 | 6.8 |
| 11/03–17/03 | 7.5 | 13.6 |
| 17/03–27/03 | 35 | 46.6 |
| 27/03–29/03 | 2.5 | 0.4 |
| 29/03–30/03 | 10 | 7 |
| 30/03–17/04 | 55 | 71.2 |
| 17/04–18/04 | <1 | 1 |
| 18/04–23/04 | 11 | 10.6 |
| 23/04–24/04 | 2.5 | 4.8 |
| 24/04–25/04 | 1.5 | 3.2 |
| 25/04–26/04 | <1 | 3.4 |
| 26/04–27/04 | <1 | 1.4 |
| 27/04–11/05 | 75 | 63.6 |
| 11/05–12/05 | <1 | 1.8 |
| 12/05–13/05 | <1 | 2 |
| 13/05–14/05 | <1 | 1.4 |
| 14/05–18/05 | 10 | 12.8 |
| 18/05–02/06 | 18 | 60.6 |
| 02/06–09/06 | 25 | 26.8 |
| 09/06–16/06 | 25 | 15.8 |
| 16/06–17/07 | 28 | 70.6 |
| 17/07–12/08 | 32.5 | 34.4 |
| 12/08–13/08 | <1 | 1.6 |
| 13/08–14/08 | 0 | 0.2 |
| 14/08–15/08 | <1 | 0 |
| 15/08–20/08 | 12.5 | 10.6 |
| 20/08–14/09 | 27.5 | 17.6 |
| 14/09–17/09 | <1 | 3.6 |
| 17/09–18/09 | <1 | 0.4 |
| 18/09–19/09 | <1 | 2 |
| 19/09–20/09 | <1 | 1.4 |
| 20/09–16/10 | No data | 33.1 |
| 16/10–17/10 | <1 | 3.4 |
| 17/10–18/10 | <1 | 0.6 |
| 18/10–19/10 | 20 | 15 |
| 19/10–22/10 | 20 | 12.4 |
| 22/10–06/11 | 11 | 19.6 |
| 06/11–07/11 | <1 | 2.4 |
| 07/11–04/12 | 70 | 48 |

rainfall (table 13) between these two localities, although they are only 15 km apart on the east coast (fig. 1). For example, over a 15-day period in May 1976, 18 mm of rain fell at Green Gorge but 60.6 mm fell at the station, and in four days in October 1976 Green Gorge had 20 mm whereas the station had only 13 mm. In 19 corresponding 24-hour rainfall records available from both localities the maximum discrepancy was 3 mm with a total of 10 mm difference over the 19 days. However, the station rainfall records corresponded with each burrow flooding event observed, irrespective of where on the island this had occurred.

If the minimum amount of rainfall required to cause burrow flooding is actually about 6 mm in the preceding 24-hour period, then, on average, this will occur four times per month. However, the frequency of 6 mm or more of rain falling within a 24-hour period is increasing. The frequency of 24-hour periods within which 12 mm or more of rain falls has also increased. While the total annual rainfall figures for the years between 1968 and 2006 indicate a general increase (fig. 2), it would appear that the greatest contribution to this is from the amount and frequency of higher rainfall quantities within any 24-hour period (table 14). When daily rainfall is divided into four categories: days of no rain, days with insufficient rainfall to cause burrow flooding (0.2–5.9 mm), days where burrow flooding may occur (6–11.9 mm) and days when burrow flooding is most likely to occur (>12 mm), the possibility of burrow flooding occurring happens with greater regularity. In addition, the likelihood of burrow flooding is aggravated by an increasing trend of consecutive heavy rain days sufficient to cause flooding (table 15, fig. 3).

A division of the 1968–2006 period into nine-year intervals (1971–2006) indicates an apparent increasing trend for both 6–11.9 mm and >12 mm of daily rainfall events, but no clear trend for the 0.2–5.9 mm category (table 16). To ascertain which rainfall trend was likely to have the greatest impact on burrow flooding this same division of rainfall data was examined for two seasons, from November–April (greatest impact on breeding success due to burrow flooding) and from May–October (least impact on breeding success due to burrow flooding). While rainfall sufficient to cause burrow flooding (more than 6 mm/24 hours) has increased in both summer and winter, it is more pronounced (fig. 4) at a time when burrow flooding would have least impact on petrel breeding success. However, in the higher daily rainfall categories the number of days, as well as the quantity of rain, is increasing. The number of days with zero rainfall has declined (table 16, fig. 5) together with a decreasing frequency of consecutive zero rainfall days

**TABLE 14**
**Annual total rainfall on Macquarie Island between 1968 and 2006 showing the 24-hour amounts in three categories and this as a proportion of the total**

| Year | Total | 0.2–5.9 mm/24 hrs | | 6–11.9 mm/24 hrs | | 12+ mm/24 hrs | |
|---|---|---|---|---|---|---|---|
| | mm | mm | % of total | mm | % of total | mm | % of total |
| 1968 | 1079 | 544 | 50.4 | 297 | 27.4 | 238 | 22.1 |
| 1969 | 944 | 484 | 51.3 | 212 | 22.5 | 248 | 26.3 |
| 1970 | 818 | 506 | 61.9 | 187 | 22.9 | 125 | 15.3 |
| 1971 | 789 | 395 | 50.1 | 220 | 27.9 | 174 | 22.1 |
| 1972 | 820 | 505 | 61.6 | 256 | 31.2 | 59 | 7.2 |
| 1973 | 766 | 350 | 61.3 | 106 | 22.7 | 311 | 16 |
| 1974 | 827 | 446 | 53.9 | 191 | 23.1 | 190 | 23 |
| 1975 | 900 | 490 | 54.4 | 272 | 30.2 | 138 | 15.3 |
| 1976 | 941 | 462 | 49.1 | 261 | 27.7 | 218 | 23.2 |
| 1977 | 946 | 514 | 54.3 | 274 | 29 | 158 | 16.7 |
| 1978 | 1013 | 561 | 55.4 | 247 | 24.4 | 204 | 20.1 |
| 1979 | 854 | 486 | 56.9 | 283 | 43.1 | 86 | 10.1 |
| 1980 | 881 | 488 | 55.4 | 203 | 23 | 190 | 21.6 |
| 1981 | 760 | 437 | 57.5 | 164 | 21.6 | 159 | 20.9 |
| 1982 | 1055 | 496 | 40 | 360 | 34.1 | 198 | 18.8 |
| 1983 | 1005 | 463 | 46.1 | 373 | 37.1 | 169 | 16.8 |
| 1984 | 1028 | 508 | 49.4 | 266 | 25.9 | 254 | 24.7 |
| 1985 | 823 | 439 | 53.3 | 254 | 30.9 | 130 | 15.8 |
| 1986 | 955 | 524 | 54.9 | 249 | 26.1 | 182 | 19.1 |
| 1987 | 931 | 531 | 57 | 258 | 27.7 | 142 | 15.3 |
| 1988 | 1156 | 493 | 42.6 | 432 | 37.4 | 230 | 9.9 |
| 1989 | 911 | 469 | 51.5 | 319 | 35 | 124 | 13.6 |
| 1990 | 1079 | 518 | 48 | 346 | 32.1 | 215 | 19.9 |
| 1991 | 1002 | 541 | 54 | 255 | 25.5 | 206 | 20.6 |
| 1992 | 1113 | 495 | 44.5 | 376 | 33.8 | 242 | 21.7 |
| 1993 | 1188 | 520 | 43.8 | 367 | 30.9 | 301 | 25.3 |
| 1994 | 1080 | 535 | 49.5 | 264 | 24.4 | 281 | 26.1 |
| 1995 | 1073 | 517 | 48.2 | 309 | 28.8 | 147 | 23 |
| 1996 | 984 | 513 | 52.1 | 349 | 35.5 | 123 | 12.5 |
| 1997 | 945 | 562 | 59.5 | 241 | 25.6 | 142 | 15 |
| 1998 | 1067 | 530 | 49.7 | 381 | 35.7 | 157 | 14.7 |
| 1999 | 1074 | 590 | 54.9 | 264 | 24.6 | 219 | 20.4 |
| 2000 | 1189 | 576 | 48.4 | 390 | 32.8 | 223 | 18.8 |
| 2001 | 1069 | 541 | 50.6 | 313 | 29.3 | 215 | 20.1 |
| 2002 | 1023 | 514 | 50.2 | 285 | 26.7 | 224 | 21.9 |
| 2003 | 1030 | 498 | 48.3 | 282 | 27.4 | 250 | 24.3 |
| 2004 | 1009 | 455 | 45.1 | 322 | 31.9 | 232 | 23 |
| 2005 | 1244 | 500 | 40.2 | 371 | 29.8 | 374 | 30.1 |
| 2006 | 1094 | 569 | 52 | 310 | 28.3 | 215 | 19.7 |

**TABLE 15**
**The risk of increased burrow flooding from the frequency of consecutive 24-hour periods of >6mm rainfall between 1968 and 2006 on Macquarie Island**

| Year | Number of consecutive days | | | | Total |
|---|---|---|---|---|---|
| | 2 | 3 | 4 | 5 | |
| 1968 | 4 | 1 | 0 | 0 | 5 |
| 1969 | 5 | 0 | 0 | 0 | 5 |
| 1970 | 2 | 0 | 0 | 0 | 2 |
| 1971 | 0 | 0 | 0 | 0 | 0 |
| 1972 | 2 | 1 | 0 | 0 | 3 |
| 1973 | 2 | 1 | 0 | 0 | 3 |
| 1974 | 2 | 0 | 0 | 0 | 2 |
| 1975 | 2 | 2 | 0 | 0 | 4 |
| 1976 | 4 | 1 | 0 | 0 | 5 |
| 1977 | 4 | 0 | 0 | 0 | 4 |
| 1978 | 1 | 1 | 0 | 0 | 2 |
| 1979 | 1 | 0 | 0 | 0 | 1 |
| 1980 | 1 | 0 | 0 | 0 | 1 |
| 1981 | 2 | 0 | 0 | 0 | 2 |
| 1982 | 3 | 2 | 0 | 0 | 5 |
| 1983 | 7 | 1 | 0 | 0 | 8 |
| 1984 | 5 | 1 | 0 | 0 | 6 |
| 1985 | 0 | 0 | 0 | 0 | 0 |
| 1986 | 7 | 2 | 0 | 0 | 9 |
| 1987 | 4 | 0 | 0 | 0 | 4 |
| 1988 | 10 | 0 | 0 | 0 | 10 |
| 1989 | 6 | 1 | 0 | 1 | 8 |
| 1990 | 4 | 3 | 0 | 0 | 7 |
| 1991 | 7 | 0 | 0 | 0 | 7 |
| 1992 | 7 | 1 | 0 | 0 | 8 |
| 1993 | 6 | 0 | 0 | 0 | 6 |
| 1994 | 2 | 1 | 0 | 0 | 3 |
| 1995 | 6 | 1 | 0 | 0 | 7 |
| 1996 | 5 | 2 | 0 | 0 | 7 |
| 1997 | 3 | 0 | 0 | 0 | 3 |
| 1998 | 3 | 1 | 0 | 0 | 4 |
| 1999 | 1 | 0 | 0 | 0 | 1 |
| 2000 | 6 | 1 | 0 | 1 | 8 |
| 2001 | 8 | 0 | 0 | 0 | 8 |
| 2002 | 2 | 1 | 0 | 0 | 3 |
| 2003 | 6 | 0 | 1 | 0 | 7 |
| 2004 | 3 | 3 | 0 | 0 | 6 |
| 2005 | 7 | 1 | 0 | 0 | 8 |
| 2006 | 5 | 1 | 0 | 0 | 6 |

**TABLE 16**
**Average number of days per year with different levels of rainfall for four nine-year blocks from 1971–2006 on Macquarie Island**

| 9-year intervals | Rainfall amount categories | | | |
|---|---|---|---|---|
| | 0 rainfall days | 0.2–5.9 mm days | 6–11.9 mm days | >12 mm days |
| 1971–1979 | 62.4 | 261.3 | 32.3 | 8.9 |
| 1980–1988 | 61.8 | 257.9 | 37.1 | 10.3 |
| 1989–1997 | 44.0 | 272.1 | 38.9 | 11.7 |
| 1998–2006 | 39.7 | 270.8 | 41.1 | 13.4 |

(fig. 6). While the average number of days per month of zero rainfall is higher when there is greater potential benefit of this to rabbit breeding success (summer), only occasionally (fig. 9) are there enough days of zero rainfall to have any influence. Such benefit would, even so, be dependent on the pattern of heavy, burrow-flooding rainfall in relation to such dry periods.

### Impacts of wind direction, wind speed, temperature and rainfall trends on burrow-nesting petrels

Changes to weather factors such as prolonged high or low temperature, wind direction and speed, in association with altered rainfall pattern, are believed to been responsible for the following:

- by 1999, rat and mouse abundance had increased dramatically on Macquarie Island, based on the frequency of sightings;
- by 1999, rats unexpectedly disappeared from one traditionally heavily rat-impacted offshore stack petrel colony;
- by 1999, rat distribution and abundance expanded with an apparent increase in availability of food such as *P. hookeri* seeds;
- an apparent change in conditions in shoreline beach strata leading to mice thriving on a proliferation of kelp fly (*Coelopa curvipes* Hutton, 1902 and *C. nigrifrons* Lamb, 1909) larvae which mice were seen eating in 1999; and
- by 1999, there was a near-total dieback of *C. plumosa*, which provides protective vegetative cover against petrel burrow habitat deterioration and petrel predation, as well as an alternative food alleviating rat predation on petrels at offshore stack colonies.

Prior to 1996, Adamson *et al.* (1988) determined that northerly winds were becoming more prevalent at Macquarie Island, and there is sufficient evidence to indicate that this trend is continuing. The number of occasions in each 24-hour period for which the three-hourly wind direction between 0°and 45° was recorded indicates an increase not only in frequency (fig. 7) but also in duration (table 17). However, this same trend is not apparent at any level of daily persistence or annual frequency, if the wind direction is between 315° and 45° (fig. 8).

## DISCUSSION

### Burrow-nesting petrels and vertebrate pests

The vertebrate pest species established on Macquarie Island have had and continue to have a wide range of impacts on burrow-nesting petrel species. Feral Cats and Black Rats prey directly on eggs, young or adult birds, as did Weka, whilst European Rabbits denude vegetation and alter burrow conditions and availability. In doing so, they can expose burrows more to predation by skuas.

The consequences to burrow-nesting petrels of fluctuations in feral pest populations, changing interactions between species, and complex environmental changes were examined in this study. Observations of the responses of burrow-nesting petrels varied with the efforts and sequence of feral pest control, and were defined by the timing intervals of the monitoring study. During the period considered in this paper, Weka and cats were eliminated and there was a period of decreased rabbit population in response to the Myxoma virus.

Monitoring sites selected were ones occupied in reasonable abundance; therefore if predator activity was centred on areas with less abundant populations but with higher nesting densities, whole island population estimates extrapolated from transect data may have been affected. This may have been the case for Antarctic Prions and White-headed Petrels that had higher proportions of their populations at more widespread sites and at lower burrow densities than those at the monitoring sites. The overall response of petrels to most pest species is best represented by the population trends of Sooty Shearwaters (despite their discreet burrow concentrations), because of their wide distribution around the entire island. All other petrel species have not yet increased appreciably above population estimates indicated by Brothers (1984) and Brothers & Ledingham (2008) despite having expanding nesting ranges.

With fewer rabbits present, Antarctic Prions have encountered decreased skua predation. With cats absent, Antarctic Prions should now become more abundant, despite the continued presence of rodents on Macquarie Island.

Sooty Shearwaters which breed in small, widely-distributed colonies in tussock, are more vulnerable to pest species than prions, especially to the impact of rabbits, and were extremely vulnerable to predation by cats. The entire population remained at risk to Weka until at least 1989. Removal of these pest species leaves Sooty Shearwaters accessible to an increasing abundance of rats. Sooty Shearwater

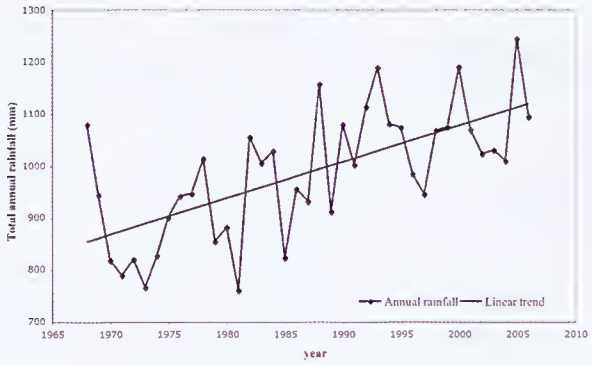


*FIG. 2 — Trend in total annual rainfall from 1968–2006 from Meteorological Station data on Macquarie Island. (Linear trend line significantly different from zero p<0.001)*

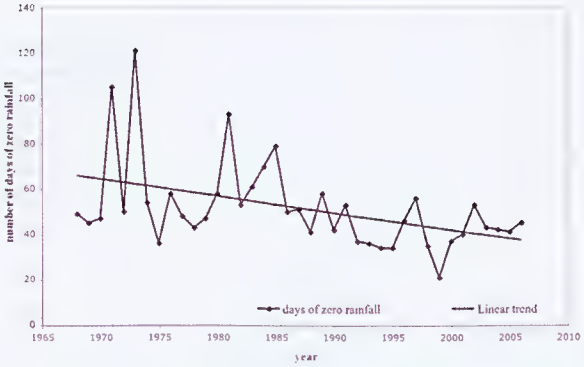


*FIG. 5 — The number of zero rainfall days on Macquarie Island from 1968 to 2006. (Linear trend line is significantly different from zero p=0.005)*

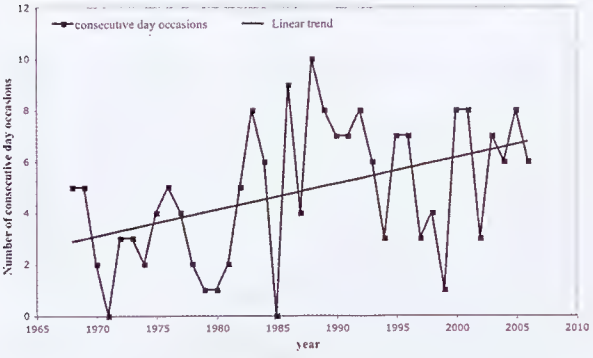


*FIG. 3 — Between 1968 and 2006, the trend in the frequency of consecutive 24-hour periods during which sufficient precipitation occurred to cause burrow flooding on Macquarie Island. (Linear trend line significantly different from zero p=0.006)*

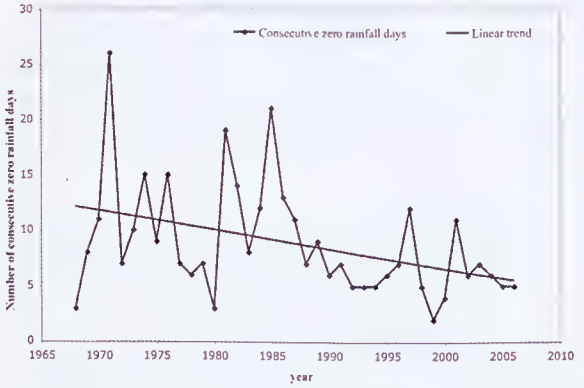


*FIG. 6 — The frequency >2 consecutive days of zero rainfall between on Macquarie Island 1968 and 2006. Linear trend line is significantly different from zero p=0.01*

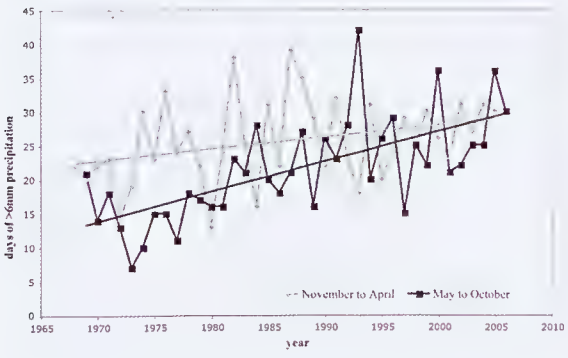


*FIG. 4 — The annual amount of heavy rain (>6mm per 24 hrs) at Macquarie Island 1968–2006 divided into greatest (summer: November 4–April) and least (winter: May–October) potential impact on breeding performance of burrow nesting petrels. (Linear trend lines show strong evidence that the rate of increase in the winter season is significant different from zero p=0.006 and is greater than in summer p=0.05.*

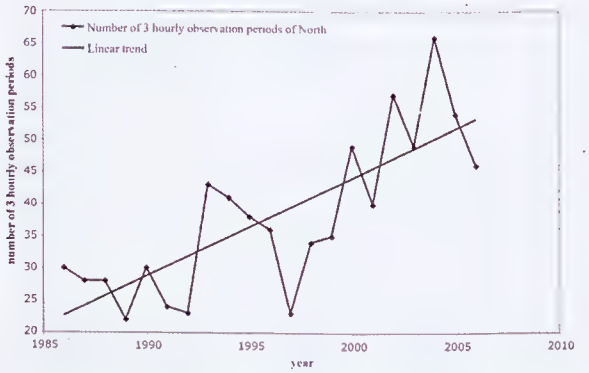


*FIG. 7 — Trend in duration of wind from 0-45° daily from 1986 to 2006 on Macquarie Island, based on three-hourly observations. (Linear trend line is significantly different from zero p< 0.001)*

*FIG. 8 — Total number of three-hourly observation periods annually in which wind direction between 315 and 45 degrees was recorded at Macquarie Island between 1986 and 2006. (Linear trend line observed slope greater than zero but not significantly different from zero p=0.5)*

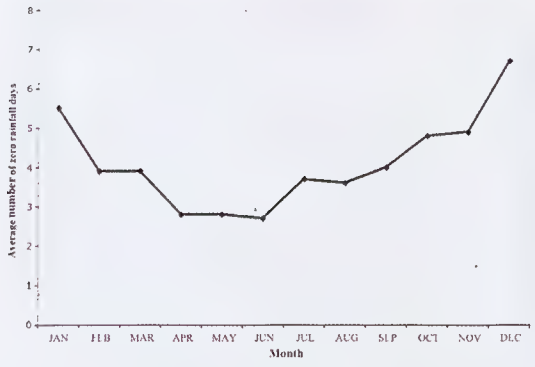


*FIG. 9 — The average number of zero rainfall days in each month on Macquarie Island between 1968 and 2006.*

**TABLE 17**
**Duration of wind from 0–45° direction each day from 1986–2006 on Macquarie Island based on three-hour observations**

| Year | Number of 3-hour records per day of 0–45° wind direction | | | | | | | | Total |
|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | |
| 1986 | 22 | 9 | 1 | 0 | 0 | 0 | 0 | 0 | 32 |
| 1987 | 25 | 4 | 3 | 0 | 0 | 0 | 0 | 0 | 32 |
| 1988 | 23 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 28 |
| 1989 | 16 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 22 |
| 1990 | 28 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 30 |
| 1991 | 20 | 3 | 1 | 0 | 0 | 0 | 0 | 0 | 24 |
| 1992 | 16 | 4 | 1 | 2 | 0 | 0 | 0 | 0 | 23 |
| 1993 | 30 | 11 | 1 | 1 | 0 | 0 | 0 | 0 | 43 |
| 1994 | 31 | 7 | 2 | 1 | 0 | 0 | 0 | 0 | 41 |
| 1995 | 22 | 4 | 6 | 1 | 2 | 0 | 0 | 0 | 35 |
| 1996 | 27 | 4 | 4 | 1 | 0 | 0 | 0 | 0 | 36 |
| 1997 | 12 | 7 | 2 | 1 | 1 | 0 | 0 | 0 | 23 |
| 1998 | 19 | 11 | 3 | 1 | 0 | 0 | 0 | 0 | 34 |
| 1999 | 21 | 7 | 5 | 2 | 0 | 0 | 0 | 0 | 35 |
| 2000 | 22 | 15 | 4 | 4 | 3 | 1 | 0 | 0 | 49 |
| 2001 | 22 | 12 | 12 | 5 | 0 | 0 | 1 | 0 | 52 |
| 2002 | 21 | 14 | 9 | 6 | 2 | 2 | 2 | 1 | 57 |
| 2003 | 22 | 10 | 8 | 2 | 5 | 2 | 0 | 0 | 49 |
| 2004 | 29 | 7 | 14 | 8 | 4 | 3 | 1 | 0 | 66 |
| 2005 | 32 | 9 | 8 | 3 | 1 | 1 | 0 | 0 | 54 |
| 2006 | 25 | 14 | 5 | 1 | 1 | 0 | 0 | 0 | 46 |

population trends clearly show the relationship between rabbit population control and a reverse in the shearwater population decline.

For the proportion of White-headed Petrel colonies still within tussock, destruction by rabbits has been highly erratic over time and patchy in extent. This habitat degradation has exposed this species to skua predation and the likelihood of burrow collapse and more regular flooding.

White-headed Petrels spend considerable time in aerial courtship whereas Sooty Shearwaters are reliant on surface courtship activity for which concealment by vegetation would be advantageous. For this reason Sooty Shearwaters are less resilient to the impact of rabbit destruction of colony habitat. There are few instances of shearwaters breeding where there used to be tussock whereas White-headed Petrels persisted and have done so long after the tussock has been removed from around burrows.

During the period of low rabbit abundance prior to 1996 when eradication effort intensified (Copson 2004), cats probably reduced White-headed Petrel numbers significantly and were largely responsible for population decline throughout the monitoring period. Threatened Species Section (2007) mentioned a small increase in this species' abundance through the monitoring period, but this was erroneous, being based on a tentative estimate (Brothers 2000). Recent escalation in rabbit damage in areas previously little affected (A.Terauds pers. comm.) could foreshadow a significant decline in White-headed Petrel numbers. Hill 291 has been bare of tussock for at least 25 years and, despite this, White-headed Petrels have continued to occupy the site with census data to 1999 indicating a small decline in numbers. Other monitoring sites for this species have been subjected to significant habitat modification by rabbits since the mid-1970s with the majority of vegetation cover that offered protection from predators now absent. However, because rats are reliant on being within, or in close proximity to, adequate vegetative cover, and cats and Weka are now gone from the island, there is an overall decrease in predator pressure. Predation by skuas advantaged by exposed burrows and surface activity of petrels may increase, but there is also the possibility that skuas may be less successful when tackling White-headed Petrels in such open ground. White-headed Petrels are large and powerful birds that can take flight more easily from open ground than from within tussock.

Because rabbits and White-headed Petrels (or other petrel species of similar size), require burrows of similar diameter, there is likely to be a direct relationship between high rabbit abundance and a decline in the population of this petrel, due to competition. In contrast, smaller species, such as the Antarctic Prion, benefit from utilising existing rabbit burrows, constructing separate nesting burrows off them. This would minimise their high vulnerability to predators through surface activity such as burrow excavation.

Brothers & Copson (1988) anticipated that, with the eradication of cats and Weka, burrow-nesting petrel populations would withstand rat predation in the long term. This hypothesis has yet to be verified. Grey-backed Storm-Petrels, *Garrodia nereis* (Gould,1841), with their small size and a preferred nesting habitat that is occupied by rats, together with almost year-round presence ashore (Marchant & Higgins 1990), are considered unlikely to regain former abundance if rats are not eliminated. Currently, this species is only suspected, not yet confirmed, to breed on the island. Rounsevell & Brothers (1984) assigned a nominal figure of breeding population size to this species based on circumstantial evidence. No further evidence of this species was encountered between 1984 and 2000.

For all other species, rats are unlikely to prevent population increase, although they may hinder total population recovery. Despite rats being still present, with cats and Weka gone, a number of species have attempted to re-establish. Such activity did not occur when cats were prevalent. Environmental factors may change the distribution and abundance of rats in the future, which may increase the threat that they pose. Rats have been observed exploiting luxuriant *P. hookeri* growth for protective cover and food (from seeds) to extend their usual distribution limit. This may be a result of more favourable climatic conditions and decreased rabbit grazing. Any such increase in food availability could allow rats to become even more prolific and an increase in their distribution would further threaten certain petrel species. The single occasion of extreme cold in 1995, suggested as responsible for the disappearance of rats, and resulting in successful Blue Petrel breeding activity in the Langdon Point offshore stack colony, does not alone explain how petrels have persisted for so long there. Examination of meteorological records suggests that the persistent cold of July 1995 has been at least equalled twice, in July 1954 and in August 1964. This low frequency of extreme cold events could be sufficient to remove rats from offshore stacks and maintain these small petrel populations. However, this is dependent on the length of time interval before the rats reinvade. This time interval has not been determined but rats were absent from the offshore stack at least five breeding seasons subsequent to the 1995 event.

Contrary to Threatened Species Section (2007), now that potential alien predators (other than Black Rat) have been eliminated, diving-petrels could establish on the island, because historically their distribution included higher altitude habitat (Campbell 1901) well outside the current and future likely range of Black Rats. Given the likely favourable response of all petrel species to future (and past) vertebrate pest eradication, the population status of petrels will require regular evaluation if these responses are to be known and reported with any reasonable degree of accuracy (Baker *et al.* 2002).

## Cats and rabbits

Cat eradication was not achieved until 1999, despite a sustained effort for over a decade (Copson 2004). In contrast to the gradual benefits from rabbits being kept in low numbers between 1984 and 2000 or from the eradication of Weka in 1989 (Copson 2004), a number of species showed immediate benefit from the rapid reduction, then eradication, of cats. Although this infers that cats were the most destructive of these pests, it was cats in combination with rabbits and Weka that were responsible for the demise of so many of the islands indigenous species. The extent to which each pest species and their combined effects contributed to the demise of indigenous species probably varied considerably over time because of the large population fluctuations of introduced species, regardless of human intervention.

Until at least 2003, rabbit numbers had effectively been maintained at around 10–15% of the pre-myxomatosis level of around 150 000 (Copson 2004). This was considered important to improve the likelihood of cats being eradicated because while rabbit numbers were contained long enough for cats to be eradicated, failure to maintain this control

has seen rabbit numbers and damage to vegetation escalate (Bryant & Shaw 2007). Ongoing control was compromised by the failure to ensure supplies of Myxoma virus to maintain an effective inoculation program. Any perception that cat eradication was somehow a catalyst for the rabbit population increase that subsequently occurred is erroneous and most convincingly disputed by the historical trends in rabbit abundance on the island irrespective of cat numbers through that time (Copson *et al.* 1981, Copson 2004). Cat predation on rabbits has never been sufficient to regulate rabbit numbers, which is not surprising considering that there were only ever known to be approximately 250 adult cats on the island (Brothers *et al.* 1985).

Feral pest management processes identified the need for cats and Weka to be eliminated so that burrow-nesting petrels could recolonise (Parks and Wildlife Service 2006). This was achieved, averting even greater impacts on remaining petrel populations. Cats themselves were believed to have been largely responsible for Weka extermination because such a rapid disappearance could not be accounted for by hunting alone (Copson 1995). Weka impact on petrels was reduced when increased rabbit grazing limited the safe foraging range of Weka. Based on the results of petrel monitoring, the prediction that petrels would face increased predation from cats as a consequence of rabbit control (Brothers 1984), seems to have occurred prior to cats being completely eradicated.

The permanent rabbit count areas (Copson *et al.* 1981) failed to accurately reflect the extent of recent rabbit increases because counting rabbits in habitat where they are less conspicuous (such as on steep tussock-covered slopes) is inaccurate. In addition, vegetation recovery resulting from reduced rabbit grazing pressure altered rabbit visibility in many count areas from 1988 to 1999, thus compromising comparisons with past counts. With cats and Weka gone, it will be important to monitor the impact of rabbits, especially at White-headed Petrel monitoring sites, which were at varying stages of measurable rabbit impact.

The relationship between rabbit and skua distribution and abundance, which in turn affects the extent of skua predation on petrels, remains complex. Rabbit control, and the concurrent contraction of skua breeding range to become more aligned with the proximity of indigenous food sources, is likely to have considerably reduced skua pressure on petrels. However, the extent of this association is unclear. While rabbit distribution and abundance prior to myxomatosis may have supported more skuas in total with nesting territories more widely distributed, it is not clear if these skuas were consuming more or fewer of the petrels nesting there. Skuas may have consumed more petrels because availability of rabbits had induced skuas to spend more time in petrel-nesting areas than usual. However, if these areas were not claimed as nesting territories by skuas, and so being held against visiting skuas, many more skuas may have been able to visit and utilise the petrel colonies as a food source.

## Recolonisations

Since the first known observation of Cape Petrels occupying Macquarie Island slopes in October 1999, there was similar activity in the subsequent two years until breeding was confirmed in 2002 (Threatened Species Section 2007). However, it is not known whether breeding occurred prior to this without being detected, particularly if breeding failed soon after egg laying. While Threatened Species Section (2007) speculates that because Cape Petrels are conspicuous they would not have been overlooked as a breeding species, Law & Burstall (1956) suspected nesting by this species in 1951 on Anchor Rock, which is relatively close to the current breeding site. Furthermore, in 1976 Lugg *et al.* (1978) suspected nesting of Cape Petrels on Anchor Rock and since then there have been frequent observations of birds alighting there. Observations of Cape Petrels on nearby Gorilla Rock have also been made. Therefore, this seems unlikely to be a case of new colonisation; more probably management actions on the island have created sufficiently safe conditions for the small numbers that have always been nesting offshore to now move to more suitable nesting sites. However, Threatened Species Section (2007) notes it as being a conspicuous and readily identifiable species, with no mention made by Hamilton (1894) or Falla (1937) of it being resident. Other surface-nesting species, the Antarctic Tern and the Kelp Gull, have been affected significantly by the presence of introduced predators to Macquarie Island with most nesting having been confined to offshore land masses with some degree of protection from predators and inundation by seas. All these species have apparently responded to the absence of cats and Weka by attempting to breed in sites where these predators would previously have had access, yet the ongoing presence of rats has not deterred them.

In addition to the impacts of introduced vertebrates on indigenous fauna on Macquarie Island, at least two species, the Macquarie Island Rail, *Rallus philippensis macquariensis* (Hutton, 1879), and the Macquarie Island Parakeet, *Cyanoramphus novaezelandiae erythrotis* (Salvadori, 1891), have become extinct (Taylor 1979). It remains to be seen whether similar species that occur on the closest landmasses become established when the opportunity arises, once the cause of their extinction on Macquarie Island, has been addressed. The Red-crowned Parakeet, *C. novaezelandiae* (Sparrman, 1787), and the Buff-banded Rail, *Rallus philippensis* (Gray, 1843), from New Zealand populations are likely candidates. However, during 25 years of vagrant bird species observations, no records of parakeets and only one rail record, that of a Baillon's Crake, *Porzana pusilla* (Pallas, 1776), have been made (Copson & Brothers 2008). It is unknown whether the Macquarie Island Rail occupied a similar ecological niche to the Weka but they probably would have had some feeding habits in common, including predation of petrels. In the absence of both species, it is possible that certain species, such as the orchid, *Nematoceras dienemum* (D.L.Jones)D.L.Jones,M.A.Clem et Molloy, bulbs of which were sought after by Weka, will increase in distribution or abundance, just as would *N. sulcatum* M.A.Clem et D.L.Jones. It is also possible that species such as the Common Blackbird, *Turdus merula* Linnaeus, 1758, and Song Thrush, *Turdus philomelos* C.L. Brehm, 1831, which have been infrequent vagrants on Macquarie Island, will establish and their impacts may warrant management consideration. The Common Starling, *Sturnus vulgaris* Linnaeus, 1758, is well established. However, unlike the Common Blackbird and Song Thrush, Common Starlings have managed this, despite predators, probably because their breeding sites are often on inaccessible cliff crevices and because they largely feed in open areas. Both the Song Thrush and the Common Blackbird confine themselves within or adjacent to tall tussock grassland where, at least in the past, their existence has been short-lived probably due to predators. Surprisingly, the Redpoll *Carduelis*

*flammea* (Linnaeus, 1758), managed to establish despite being vulnerable to indigenous predators (gulls and skuas) and to all vertebrate pests. Undoubtedly, species with a flocking habit such as starlings and Redpolls will colonise new places more readily by arriving in numbers. Therefore, it is considered that the Red-crowned Parakeet would precede other colonising species.

To date, management response to discovery of alien plant species, thought to be recent arrivals to the island, has been to exterminate them (Copson & Whinam 2001) and Brothers & Davis (2008) discuss the issue in relation to alien marine life encountered. This needs to be further considered in relation to the avian arrivals that will establish in the absence of, in particular, Weka and cats. The likelihood of new bird species establishing on the island may have increased due to climate change, as there have been changes in wind directions more favourable for southward movement (Adamson *et al.* 1988). This wind direction trend is continuing (fig. 7, table 17).

## Ongoing problems with rats

Copson (2004) outlined a localised rat eradication and control program, initiated as part of the petrel monitoring work in November 1999 to coincide with eradication of Weka and cats. At this time, there was increased petrel burrowing activity coupled with, possibly, drier conditions less favourable for maintaining *C. plumosa* vegetative cover and soils resistant to skua excavation. This resulted in poor petrel-breeding performance and, therefore, impeded recolonising of Macquarie Island. With an immediate objective of rat eradication from all offshore stacks, ongoing control on the shore adjacent to stacks was instigated using a series of rat poison bait stations in an effort to both prevent reinvasion and to create rat-free nesting habitat to which petrel populations could expand. This would also help create a situation where predation by rats would be less devastating to petrel populations and so allow petrel populations to increase. Coincidentally, examples of the viability of such a strategy had been illustrated by the fact that petrels had already recolonised Macquarie Island via the only rat-free offshore stack. Data from the Langdon Point colony indicate that keeping stacks rat-free to improve petrel-breeding performance is viable. Jouventin *et al.* (2003) also provide evidence of the benefit to petrel-breeding performance through maintaining population control of rats. However, the perceived risk of poison immunity developing in the rat population prior to implementation of the rat eradication plan was the reason that localised rat control for petrel colonies ceased in 2003 (Copson 2004). Maintaining this effort specifically on offshore stacks should have posed minimal risk of poison immunity, but there was also uncertainty as to whether appropriate effort could be directed at keeping the offshore stacks in a rat-free state. Reoccupation of Macquarie Island at Green Gorge by Blue Petrels is not only the most significant example of a small-sized petrel species returning from offshore stacks to breed but is also the first occasion of any breeding occurring at such a low altitude within the distribution of rats. This reinforces the interpretation that rats were never as significant in the demise of petrels as Weka, cats and rabbits. The dominant vegetation on offshore stacks, *C. plumosa*, which flourishes with the intensive soil disturbance of petrel burrowing activity, regresses naturally seasonally. However, the petrel-affected, well-drained soils and the effect of rat-browsing are likely to shorten the period of protective burrow cover there. Burrow collapses together with increased skua predation are inevitable consequences of this reduced cover.

Any petrel population increase resulting from rat eradication will lead to greater pressure from burrowing activity exacerbating loss of vegetative cover, deterioration in the burrowing habitat and eventual population decline on offshore stacks. The safest way to prevent this would be to have rat-free habitat onshore adjacent to the stacks created at the same time, although the recolonising already adjacent to the Green Gorge stack indicates that rats are not entirely preventing this without such management intervention. However, in the immediate future at least, rabbit grazing impacts that dictate to a large extent the distribution of rats, will assist, not impede, the rate that the smaller petrels can recolonise Macquarie Island in the absence of Weka and cats.

In recent times, and certainly by 1999, rats had been observed in much greater overall abundance on Macquarie Island (Copson, Brothers unpubl. data). Since then rats have not been prevented from occupying stacks, so it would be a wise strategy to maintain offshore stacks free of rats in the event that an island-wide eradication attempt is unsuccessful or does not eventuate for any reason.

Copson (2004) discussed the possible causes for the observed large increase in rodent (rats and mice) abundance, with global warming implicated. The long-term effects of changed utilisation by rats of *P. hookeri* (Shaw *et al.* 2005), and perhaps of other plant species as well, is complex. This is especially so with climate change adding uncertainty to subtle yet potentially significant ecological relationships on Macquarie Island, and is even more complicated if an eradication attempt (Copson 2004) leaves the island without rabbits but still with rodents. Increased rates of reproductive development in at least certain plant species together with the ameliorating climatic conditions forecast for the island (Tweedie 2000) may have already been responsible, at least partially, for increasing rodent abundance. Presumably these factors may also assist plant recovery from rabbit impacts as implied by Tweedie (2000), but the causes of such relationships are likely to remain unclear, particularly if eradication of rabbits occurs. At present, the rat's ability to have such a big impact depends on the extent of vegetation cover, which increased because rabbit numbers had been maintained at a greatly reduced level. Although such recovery of vegetation may increase rat populations and broaden their distribution to further overlap with sectors of the petrel population, an increasing abundance of alternative food for rats other than petrels (e.g., seeds) may reduce the impact of rats on petrels. Copson (2004) presented a strategy for achieving the eradication of rodents and rabbits and outlined the risks of an attempt to eradicate all species simultaneously. For the majority of burrow-nesting petrels, the eradication of rabbits but not rodents, would be a minimum acceptable target and is a sensible objective if an eradication attempt on all species cannot proceed or fails.

When the conservation status of burrow-nesting petrel species are threatened by other anthropogenic causes such as by some of the world's fisheries (Weimerskirch *et al.* 1999, de L. Brooke & Croxall 1999, Brothers *et al.* 1999, Sullivan *et al.* 2006), the added impact of rats may become more significant. The smaller petrel species which face rat predation of eggs, chicks and also adults (Warham 1990) are fortunately not doubly disadvantaged by becoming fisheries bycatch as are some of the larger petrels. However,

of the larger species now on the island, Sooty Shearwaters, Cape Petrels, particularly Grey Petrels, and the potentially recolonising species White-chinned Petrels, *Procellaria aequinoctialis* Linnaeus, 1758, are all highly vulnerable because of increased mortalities in fisheries. Grey Petrels and White-chinned Petrels in particular are susceptible to being caught on hooks in longline fisheries. They are little deterred by bird-scaring lines, aimed at excluding their access to baited hooks (Brothers *et al.* 1999), in contrast to other susceptible seabird species. In addition they are also highly proficient pursuit divers, feed throughout both day- and night-time, are unafraid to forage persistently immediately alongside fishing vessels and take advantage of ship's lighting to pursue baited hooks. These traits impose considerable limitations upon the effectiveness of all mitigation measures currently employed to prevent longline fisheries mortalities which, combined with such low abundance of these species breeding on Macquarie Island, should dictate that they are not put at further risk by any such fishing activity occurring near the island. One immense benefit of vertebrate pest management to allow recovery of such petrel species on Macquarie Island will be the added assistance this gives to the conservation of such species in the face of high fisheries mortalities.

## Climate change impacts

It is likely that, as species now adapted to one of the worlds' most equitable climates, indigenous residents of Macquarie Island will be particularly susceptible to even relatively small changes in the climatic conditions. This is becoming increasingly apparent for vertebrate species, through the impacts of climate change on habitat. The timing of this has complicated the accurate interpretation of management actions and their impact. For example, the dramatic increase in the abundance of rodents is coincidental with, but not necessarily a consequence of, cat eradication, because cats were insufficiently abundant (Jones 1977, Brothers *et al.* 1985) and had adequate alternative food supplies. If nothing else, this should reinforce the necessity for planned management of vertebrate pests to proceed, preferably sooner rather than later, in order to avert rapid progression of rabbit and rodent impacts on the island under climatic conditions more suited to these species. Indications of relationships discussed between specific climatic conditions and fluctuations in pest species abundance need to be taken into consideration with respect to the proposed eradication plan. The question of whether climatic conditions for greatly improving eradication prospects can be predicted from the island's long-term meteorological records needs to be answered. Such specific relevant climatic conditions have been identified here and those conditions more suited to the logistic requirements of an eradication attempt (Copson, 2004) are also identifiable. Aside from the complications of a climate increasingly more suited to proliferation of the remaining vertebrate pests, there is the possibility that eradication of the three remaining vertebrate pests may be unachievable. The significance, therefore, of relationships such as that between precipitation patterns and water table movement must not be underestimated with respect to their impacts now and in the future on both introduced and indigenous species on the island. A more detailed study of this relationship is needed together with combined effects of other weather features that are also changing (Adamson *et al.* 1988, Jacka *et al.* 2004, Pendlebury & Barnes-Keoghan 2007).

The perceived heavier rainfall, soil erosion and frequency of petrel burrow flooding events have been supported by evidence of increasing heavier rainfall. More frequent flooding of rabbit burrows has probably also occurred and would account for the recent tendency for the highest rabbit abundance and resulting habitat destruction to be the greatest in well-vegetated steep coastal localities (i.e., less flood-prone habitat). More frequent heavier rainfall (sufficient to cause rabbit burrow flooding) is likely to cause altered, topographically-related rabbit population increase, because rabbit kitten mortality due to wet burrows will be greatest throughout much of the traditionally occupied rabbit habitat. On well-vegetated steep slopes that are considered less susceptible to burrow flooding and which have protection from the impact of the combination of cold and moisture on kitten survival, the population increase in the absence of myxomatosis control inevitably led to the substantial increase in habitat destruction here. There is no convincing evidence to support alternative hypotheses such as increasing immunity of rabbits to myxomatosis or prolonged breeding in response to more favourable winter weather conditions (Parks and Wildlife Service unpublished records).

While Tweedie (2000) found mean monthly precipitation to be decreasing mainly between late autumn and early spring, no evidence has been presented to indicate greater breeding performance of rabbits outside their usual breeding timetable (Skira 1978, 1980). Of further relevance may be an evaluation of the 24-hour rainfall pattern in the 0.2 mm–5.9 mm range. With an average of 266 days per year (1968–2006 meteorological data) upon which such rainfall occurs, most ecological impact may be a consequence of a change occurring in the daily pattern only of this amount of precipitation. In relation to rabbit kitten survival alone, more constantly wet, but not necessarily flooded, burrows may be of greatest relevance. Even relatively small amounts of precipitation within shorter timeframes daily could affect kitten survival. However, a more precise understanding of the relationship between rainfall amounts and water table response would be required to investigate this. While this does support observations such as those of increases in burrow flooding events and erosion rates, other weather-related changes indicated are not so readily explained. In some months, for example, there can be as many as 17 days of no rainfall that may have short-term ecological impact. It was unexpected, however, to find that there was not a trend towards more consecutive zero rainfall days each year with the island thus experiencing greater extremes of dryness as well as wetness. Further, in relation to the observations of dry condition impacts, Whinam & Copson (2006) discuss this concerning sphagnum moss, and Pendlebury & Barnes-Keoghan (2007) propose that, despite evidence of actual weather conditions, evaporation as a consequence of higher wind speeds over the island may be responsible. Relative humidity is decreasing on the island which may counterbalance the increased net radiation and precipitation occurring, and recharge of soil moisture may be decreasing (Tweedie 2000). Such a process could account, therefore, for the visual evidence described of drier conditions irrespective of precipitation increase.

Previously there has been no attempt to look in detail at the nature of precipitation patterns at Macquarie Island with interpretation of meteorological records tending to focus on overall totals to give long-term trends. Tweedie (2000) suggested, however, that the overall increase in precipitation is in the form of more intense rainfall periods

indicated by increases in both maximum daily precipitation and precipitation standard deviation. Williams (unpublished BOM data analysis 1968–2005) found that the rising trend in incidence of days with low mean sea level barometric pressure, (around 970 hpa), is the reason that rainfall is increasing, i.e., higher precipitation rate is strongly correlated with lower barometric pressure at the island. The reason for this occurrence is because the island's weather is being influenced by a high-index state of Southern Annular Mode, being in a region of increasing westerly winds potentially bringing more moisture (Pendlebury & Barnes-Keoghan 2007). Wind strength and its relationship to persistence and direction were not assessed in their study and could be an important additional factor influencing the arrival prospects of terrestrial airborne species.

It is not known how petrels, for example Grey Petrels which were not found to be breeding on Macquarie Island until 1993, and other seabirds have the capacity to immediately exploit recolonising opportunities created by extermination of introduced predators. The return of Grey Petrels to Macquarie Island is believed to be due to individuals coming from elsewhere. This is supported by the very rapid population increase following the initial arrival (and discovery) of small numbers of breeding pairs on the island. Locally displaced species, such as Antarctic Terns, Kelp Gulls and perhaps Cape Petrels, only made an attempt to reoccupy new breeding sites when the opportunity, in the absence of predators, first arose.

In contrast to the population history outlined for the Grey Petrel, the Soft-plumaged Petrel is believed not to have experienced an entirely disrupted breeding association with the island in the period of cat and Weka habitation. Unlike the smaller petrel species, none of Macquarie Island's offshore stacks or islands that could have served as a refuge, contain nesting habitat suitable for these larger petrels. Reasons for the wide range of impacts caused by introduced pests on each petrel species have been discussed by Brothers (1984).

After the Grey Petrel, the White-chinned Petrels considered to have formerly been a breeding species on Macquarie Island, possibly would have been the first petrel species eliminated. The White-chinned Petrel would have been unable to persist even in low numbers because of its diurnal activity habits ashore and typically large, accessible burrows. Accounts of wild dogs as early as 1815 (Cumpston 1968) may be sufficient explanation for this species' disappearance even before it was documented as a breeding species. Whether this species, after such a long disassociation with the island, manages to re-establish will only be revealed in the future.

## Future studies

Future censuses must accommodate the fact that all prion-sized burrows may not belong to just one prion species, because other similar-sized species will begin to recolonise the main island. These may include, in potentially intermixed colonies, the Blue Petrel, Fulmar Prion, Fairy Prion, Common Diving-Petrel and South Georgian Diving-Petrel. Likewise, shearwater-sized burrows could become intermixed with the similar-sized burrows of several more species. This may mean increased difficulty in trying to ascertain which species is occupying each burrow. Drawbacks of trying to do this include the fact that it can be destructive to fragile nesting habitat, disruptive to disturbance-sensitive species, logistically impractical on a regular basis or over the same specific locality repetitively. Irrespective, this will still not account for the proportion of breeding pairs whose breeding attempt was unsuccessful or was deferred, a frequent occurrence (Waugh *et al.* 2003). Unsuccessful or deferred breeding attempts are at least two reasons why empty burrows are not necessarily occupied by breeding pairs, and why the assumption that each active burrow represents a breeding pair may not be a false assumption, as stated by Lawton *et al.* (2006). It is not known just how accurately population size can be determined, using the assumption that each active burrow represents one breeding pair. However, while even burrows that appear to be active may not actually be used for breeding in that particular year, indications of population trends should be possible from comparisons between years of active burrows counted. Measuring call playback response as a means of ascertaining burrow occupancy is efficient and benign but imprecise because not all birds that are present respond, some that do respond may not be breeding and pairs that leave the egg unattended for periods will be missed. But the results of studying specific breeding species' habits in detail to account for such variability can then be applied to broad population monitoring as a means of improving accuracy in burrow occupancy rates. Such studies, in themselves, can create disturbance-induced biases and also require considerable, long-term investment of resources so that potential high inter-annual variability can be accounted for. A major problem arises where different species occur in abundance at burrows that are indistinguishable, which is likely to become a rapidly increasing problem as species recover their former distribution and abundance on Macquarie Island. Application of techniques for determining burrow occupant identity, rates of burrow occupancy and evidence of breeding such as the infra-red camera "burrowscope" (Markwell 1997, Hamilton 1998, 2000) could assist to measure populations of petrels more accurately on Macquarie Island provided such equipment can cope efficiently with fragile burrowing habitat, a high incidence of long, interconnected, deep, muddy and often partially flooded burrows, and then differentiate species that have very similar physical characteristics.

The importance of relatively small but abundant burrow-nesting seabirds as major consumers of marine resources (Brooke 2004) is easily overlooked at Macquarie Island where their populations are presently small compared to the vast numbers of other species (e.g., penguins). This balance is anticipated to change dramatically with petrel population recovery after the fulfilment of the management objectives for vertebrate pests (Parks and Wildlife Service 2006). Therefore, the energetic requirements of future, not current, burrow-nesting petrel populations should be considered in relation to any management of marine food resources.

For all the uncertainty of the various ways in which seabirds may be affected ashore by a changing climate, factoring in such changes offshore adds substantially to the complexity of understanding causes of species population trends. For instance, the natural climatic variability and Macquarie Island's proximity to the Antarctic Polar Frontal Zone and to the Antarctic Circumpolar Current are likely to be significant influences on species, in particular, during the breeding season when foraging is restricted. Even slight variations in such influences could have major impacts. Vastly more information will be necessary to determine whether processes of climate change in marine and terrestrial environments are having greater effects than other known or more obvious influences on the abundance of species.

Removing the impact of rabbits and rats from the equation would be of immense assistance in simplifying this complex puzzle.

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# HAURAKI MAORI MATAURANGA FOR THE CONSERVATION AND HARVEST OF TITI, *PTERODROMA MACROPTERA GOULDI*

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(with one text-figure and two tables)

**Lyver, P. O'B., Davis, J., Ngamane, L., Anderson, A. & Clarkin, P.**2008 (31:x): Hauraki Maori Matauranga for the conservation and harvest of Titi, *Pterodroma macroptera gouldi*. *Papers and Proceeding of the Royal Society of Tasmania* 142(1): 149–160. ISSN 0080-4703. Landcare Research, PO Box 40, Lincoln 7640, New Zealand (PO'BL*), c/o Hauraki Māori Trust Board, PO Box 33, Paeroa, New Zealand (JD, LN, AA, PC). *Author for correspondence. Email: lyverp@landcareresearch.co.nz

Hauraki Maori traditional knowledge (which the New Zealand Maori term matauranga) concerning the harvest of Titi, Grey-faced Petrel, *Pterodroma macroptera gouldi* (Hutton, 1869), on the islands adjacent to the Coromandel Peninsula was recorded and analysed. The harvest of Titi linked Hauraki individuals to culture, ancestors, individual well-being and tribal identity. It also maintained mana (prestige), kaitiaki (environmental guardian) responsibilities and matauranga systems. Harvest tallies of Titi chicks (and number of birders) declined from 15 000 chicks (and 100–150 birders) before 1950, to 1000–1200 chicks (10–20 birders) by the late 1980s, to < 100 chicks (5–10 birders) in 2007. Decline in harvest tallies was not due solely to fewer individuals harvesting because daily catch rates per birder also declined, in some circumstances by as much as 87%, over this time. Traditional resource management strategies for sustaining Titi populations included: selection of chicks in the intermediate stage of growth allowing those in a more advanced state to escape; harvesting chicks towards the end of the adult provisioning period to minimise disturbance; creating breeding space by splitting burrows; annual rotation of harvest around islands to enhance escapement in some years; assigning partial island refuges to enhance escapement; respecting the mana and mauri (life force) of the Titi by not leaving chick remains on the islands and causing abandonment; and designating a rahui (temporary harvest prohibition) on islands to rest colonies from harvest. Indigenous knowledge can provide valuable insights into population dynamics and strategies for managing a species, as well as to prioritise research to safeguard the population, traditional knowledge and cultural well-being of the harvesting community.

**Key Words: Maori, Hauraki, Grey-faced Petrels, traditional knowledge, resource management strategies, *Pterodroma macroptera gouldi*, Titi, matauranga.**

## INTRODUCTION

Relationships between wildlife and indigenous peoples are important for subsistence economies, defining cultural identity, and providing links to history, ancestors, land, art and environmental philosophy (IIED 1994, Kirikiri & Nugent 1995, Moller *et al.* 2004). In many instances, traditional knowledge (herein referred to what Māori term *mātauranga*) underlies and guides these relationships (Lyver 2002). Unlike science-based systems, indigenous peoples' traditional knowledge depicts ecosystems as infused with spirit and life force and based upon reciprocal human–animal relationships (Krupnik & Vakhtin 1997, Tyrrell 2007). This is quite different from a scientific perspective which views ecosystems as mechanical, quantitative and quite distinct from people and feeling (Berkes 1999).

Recording traditional knowledge holds value in its own right, but can also inform research and management for a variety of ecological systems (Ohmagari & Berkes 1997, Lyver *et al.* 1999, Huntington 2000, Gilchrist *et al.* 2005). Moller *et al.* (2004) argued that traditional monitoring methods, albeit often imprecise and qualitative, are nevertheless valuable because observations are diachronic (knowledge developed over a long timeframe and from one locality), inexpensive, incorporate large sample sizes, invite the participation of harvesters as researchers, and sometimes incorporate subtle multivariate cross-checks for environmental change. Decision making about the management of wildlife populations benefits from an in-depth understanding of the species' abundance, demographics, movement and habitat use. For wildlife managers, understanding population trends is a critical first component in the sustainable management of wildlife harvests. Estimates of abundance using scientifically-based methodologies can often be expensive and time-consuming, frequently lack replication and have observer bias, often be run for less than a single generation of the species concerned, and often be only "relative indices" or accompanied by large confidence intervals (Moller 1996, Moller & Raffaelli 1998, Raffaelli & Moller 2000). For many wildlife populations there are often insufficient quantitative data (e.g., adult survival, immigration rates) to support scientific tools (e.g., trophic or stage-structured population models) and decision making. As a result, management decisions can be made in the absence of data, or by using data from related species to fill knowledge gaps. For many wildlife management challenges, and especially those involving indigenous people, there will be little robust scientific information available to guide sustainable harvesting. Also, indigenous communities will have little access to, or even little understanding of, the value of scientific information that could secure or enhance customary life-ways. Equally many wildlife scientists will have little appreciation of the power and utility of mātauranga.

Some indigenous knowledge systems have been eroded over the past 100–200 years. New Zealand Māori recognise that there has been a breakdown in transmission of mātauranga because of cultural assimilation with European culture (Tau 2001). Separation from natural resources through government land confiscations and harvest prohibitions, motivated by a predominantly preservationist conservation paradigm, has accelerated this loss. Even so, many *iwi* (tribes) assert that their mātauranga, which is largely based around an ethic of "resource conservation for

future use" (Kirikiri & Nugent 1995, Roberts *et al.* 1995, Moller 1996), can reliably inform wildlife management and conservation in New Zealand (Lyver 2002). The value of and need to include mātauranga has been supported and ratified under international agreements such as the Convention on Biological Diversity 1993 (UNEP 1993), to which New Zealand is a signatory.

Our study considered Hauraki mātauranga as a source of information to guide the harvest of a seabird breeding on New Zealand's offshore islands. For the Hauraki iwi, mātauranga forms the basis of their association with a culturally significant seabird species, the Tītī, Grey-faced Petrel, *Pterodroma macroptera gouldi* (Hutton, 1869). The customary take of Tītī chicks by Hauraki represents one of the three remaining petrel harvests permitted in Australasia (Skira 1990, Kitson 2004).

Tītī breed mainly on islands off the east coast of northern New Zealand, with the larger colonies on Rimariki Islands, Taranga (Hen Island), Mokohinau Islands, Ruamaahua (Aldermen) Islands, Moutohora (Whale Island) and Whakaari (White Island) (Wodzicki & Robertson 1959, Imber 1976, Imber *et al.* 2000). Recent estimates of numbers on the major breeding islands suggest the population is about 200 000–300 000 breeding pairs, indicating a total population in excess of a million birds (Taylor 2000). The Tītī is unusual among *Pterodroma* because it is a winter-time breeder and lays eggs during June and July, with its chicks fledging in early summer. On average, Tītī adults are 410 mm long and weigh approximately 550 g (Heather & Robertson 1996) while their chicks usually weigh less than 900 g.

Hauraki, Ngāti Awa and Ngātiwai, and other iwi groups around the upper half of the North Island of New Zealand traditionally harvested Tītī chicks annually from mainland and offshore island breeding colonies (e.g., Rimariki Islands, Ruamaahua Islands, Moutohora Island). However, in response to declining harvests during the 1960s Hauraki and Ngāti Awa implemented a *rāhui* (temporary harvest ban) and the harvest on their islands was mostly discontinued. Even so, a small-scale take persisted from the Ruamaahua Islands by a few individuals, and now there is a renewed interest by the iwi to reinstitute the customary practice.

The objectives of this paper were to understand the significance of Tītī and its harvest to Hauraki and determine long-term changes in the abundance and density of Tītī on their islands. We also reviewed *kaitiakitanga* (environmental guardianship) strategies used by Hauraki to manage the sustainability of their harvests. We argue that Hauraki did not harvest indiscriminately, but had ways of proactively managing the Tītī population and their breeding islands. Finally, we consider the role of Hauraki mātauranga in informing wildlife and harvest management for the Ruamaahua Islands.

## METHODOLOGY

### Interview process

An invitation to conduct this research was extended by the Ruamaahua Islands Trust, and sanctioned by the governing Hauraki Māori Trust Board and community at tribal meetings in Paeroa. Selection of interviewees was deliberately non-random. Those individuals who were approached for interviews were recognised by the community to have knowledge and experiences relating to the Tītī. Therefore, interviews were conducted with seven *kaumātua* (male elders; age range 50–88 years) of which five had >30 years birding experience; four *kuia* (female elders; age range 65–78 years); four birders with less than three years' harvest experience; and four Ruamaahua Islands Trust and working group members. Each of the interviewees had either direct or indirect involvement with Tītī through the historical or recently reinitiated harvest on the Ruamaahua Islands (fig. 1) or through the current co-management of the islands with Department of Conservation (DOC). This either exposed interviewees directly to the birds, the islands, and harvest, or indirectly through knowledge collected about the system. Repeated ideas and patterns of knowledge that emerged over the course of the interviews indicated we had interviewed enough individuals for the information to be considered reliable.

Prior to commencing an interview, a project description and an oral history agreement governing information use and confidentiality were discussed with the interviewee. Semi-structured interviews, in which questions are presented in the context of discussion, were conducted to allow for a more "natural" conversation to occur and unanticipated insights to emerge (Huntington 2000, Telfer & Garde 2006). Quite often interviewees would provide responses related to a particular topic without being specifically asked about it, so the interviewer would refrain from asking those questions.

Three interviewers were used over the course of the study, although one interviewer conducted 89% of the interviews. All the interviewees spoke English as either a first or second language, so all interviews were conducted primarily in English. The interviews were conducted between 1 July 2006 and 7 November 2007 and ranged from 0.75–2 hours in length. Fourteen of the interviews were conducted on a one-to-one basis, while on four occasions discussions were conducted with two interviewees present. All of the interviews were recorded on digital video and transcribed verbatim. The accuracy of transcriptions was checked by a second transcriber. Direct quotes are indicated by indentation below.

Interview questions were developed in conjunction with the Ruamaahua Islands Trust. For the purposes of this paper the interview was divided into four broad sections and addressed key themes about: (1) the significance of Tītī and its harvest to Hauraki; (2) observed changes in Tītī abundance and variation in burrow and chick densities; (3) cultural indicators used to gauge changes in Tītī abundance; and (4) traditional kaitiakitanga strategies used to manage the Tītī harvest and islands.

## RESULTS

### Harvest of Tītī

Interviewees reported that Tītī were usually harvested during the first two weekends in November. The first birding party would go out and harvest on the opening Saturday, and the second party on the following Saturday. Harvesting did not usually occur on the Sunday, although occasionally birders would overnight on the islands and return to the mainland the following day. Tītī chicks were only harvested from burrows during the day. Interviewees reported that if a chick was close to the burrow entrance, it could easily be caught

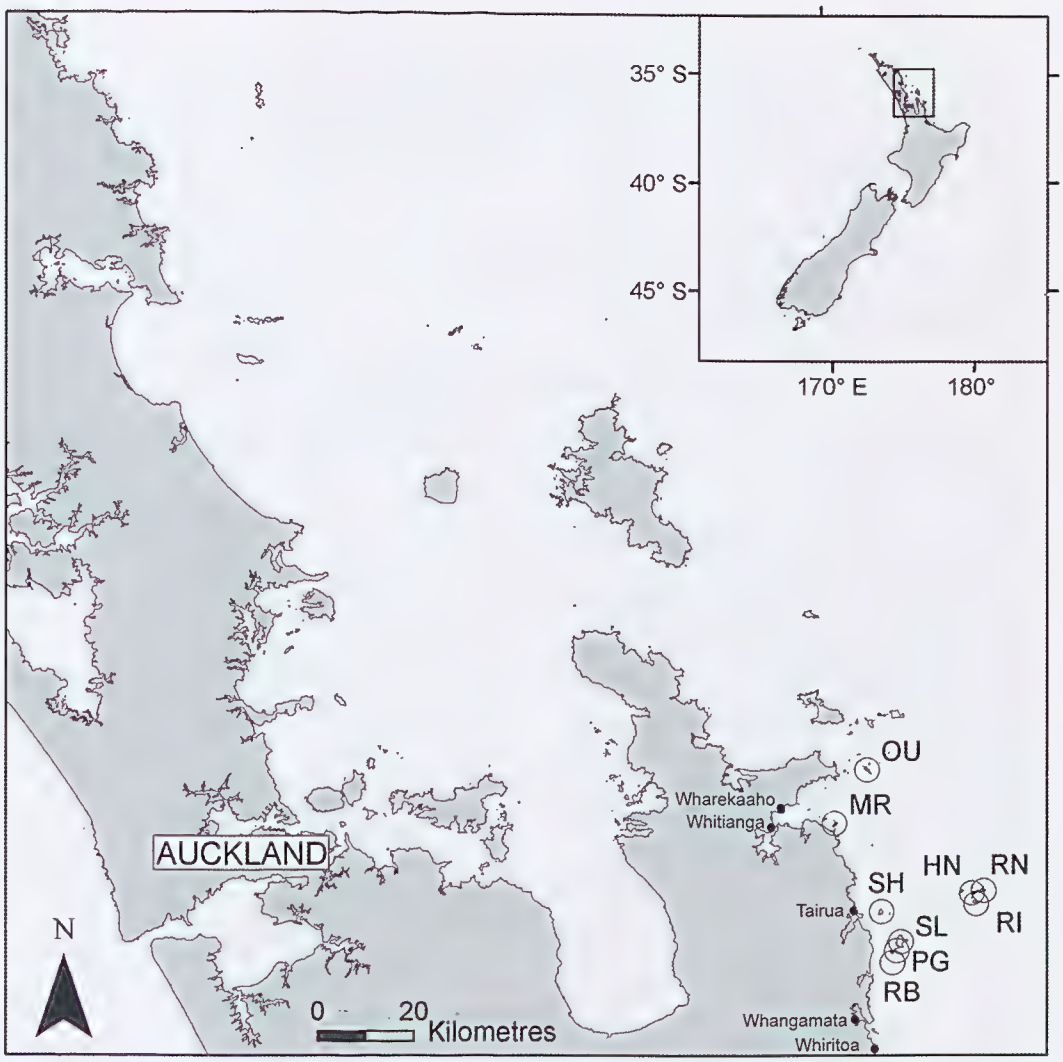


*FIG. 1 — Hauraki communities and Tītī Islands harvested by Hauraki birders (Ruamaahua Islands [RN – Ruamaahuanui; RI – Ruamaahuaiti; HN – Hongiora]; OU – Ohinau; MR – Mahurangi; SH – Shoe Island; SL – Slipper Island; PG – Penguin Island; RB – Rabbit Island).*

by hand. However, if a chick was beyond hand reach, a stick of mānuka, *Leptospermum scoparium*, J.R. Forst & G. Forst, or bracken fern, *Pteridium esculentum*, (G. Forst) Cockayne, between 0.5–1 m long with short prongs (~ 5 mm) protruding from the end would be inserted. Once birders could feel the chick pecking the stick or moving around in the burrow, they would twirl the end into the bird's down so it would become entangled. It would then be quietly eased up the burrow to a point where it could be reached and pulled out. Interviewees indicated that it was a skill that needed experience to be perfected.

Once the bird was in hand it was quickly killed by breaking its neck or crushing its skull. Before placing the dead chick into a sugar-bag, some birders would apply pressure on the bird's abdomen causing it to regurgitate its stomach contents, while others would pull or cut out its crop. Some birders indicated they just harvested the chicks and took them back to the mainland in their entirety and processed them at home.

The older birders recounted how before the 1970s, chicks were usually brought home and dipped in either the creek or a pot of hot water to assist with feather and down removal. Other birders indicated that their families would conduct the entire preparation process while out on the island. Burning candles or paper were also used occasionally by some to singe off down and pin feathers. After about 1968, many birders would partially pluck birds and then immerse them in molten wax. Once the wax had set it would be cracked and stripped away taking with it all the remaining down and pin feathers. Birds were mostly eaten fresh or frozen for later use. Only two birders remembered birds being salted or preserved in fat after especially large harvests.

## Significance of Tītī to Hauraki

Interviewees explained the importance of Tītī harvest in terms of a spiritual association that the practice provided to culture and ancestors; it also provided individual well-being and iwi identity as a way of maintaining *mana* (prestige) and *kaitiaki* (environmental guardian) responsibilities and knowledge. They also acknowledged direct benefits of physical nourishment and nutrition from eating the bird (table 1).

One interviewee reported that there was an expectation from other iwi in Hauraki that his family would provide the traditional foods like Tītī or *kaimoana* (seafood) for occasions of cultural and social significance. It was emphasised that the

**TABLE 1**
**Significance of Tītī and their harvest to Hauraki**

| No. | Explanation of significance |
|---|---|
| 1 | Desire to see Tītī (and seafood) back on the tables of our *koroua* (elder men) and *kuia* (elder women) here in Hauraki and at our *hui* (gatherings). |
| 2 | Having Tītī present on the table to eat on important occasions is very relevant for Hauraki. It partly identifies the Hauraki people or particular families within the *iwi* (tribe). It denotes where you come from. |
| 3 | Tītī are a traditional food source for Hauraki and have been a staple food item in the past. It is about continuing to have the taste of those traditional foods. |
| 4 | There is a level of expectation that certain *hapū* (subtribe) or *whānau* (family) will provide particular types of foods for occasions. There is a *mana* (authority and prestige) aspect to being able to put local wild food on the table for visitors at gatherings. |
| 5 | It is the customary right of Hauraki to harvest Tītī. Continuation of the harvest maintains that right and tradition. |
| 6 | The harvest of Tītī is a traditional practice passed down through the generations by Hauraki *tūpuna* (ancestors). Continuation of this practice maintains that link to the tūpuna. |
| 7 | Continuation of the harvest maintains the integrity of the *mātauranga* (Māori trandional knowledge) associated with the Tītī, the harvest and islands. Isolation from the harvest can disrupt the oral transfer of knowledge. The associated mātauranga is a living thing. It is an evolving thing. |
| 8 | Continuation of the harvest preserves the traditions and unity within and amongst the iwi, hapū and whānau. |
| 9 | Continuation of the harvest maintains the connection and the *kaitiaki* (guardian) relationship with the Ruamaahua Islands and the bird. |
| 10 | The Tītī is a special *kai* (food). It is a *rangatira* bird (a bird with chiefly status) and can not be replaced with other poultry like chicken. It is a special bird that nourishes not only your physical self, but also your spiritual well-being. It is a food that you crave. |
| 11 | Provision of your own food whether it is from the sea, forest or garden provides a sense of satisfaction that one does not get from store-bought food. There is a special feeling in eating the same foods that the old people ate. |

ability to supply abundant and high-quality local food was an essential part of the *manaakitanga* (hospitality) customs and responsibility:

> *Whenever there was a tangi [funeral] on our [eastern] side of the [Kaimai] ranges we never had to worry about going to gather kaimoana or Tītī. However, when there was tangi or hui [gathering] or something over on the other side, the expectation from the people over there was that we would put kaimoana or Tītī on the table. From the time I was a little boy that would always happen.* (J.L. 2007)

Interviewees recognised that the knowledge pertaining to the *tikanga* (traditions and rituals) such as *karakia* (prayer); *pātere* (rhythmical chant); *waiata* (traditional song); *pepeha* or *whakatauki* (proverbs); *tohu* (signals or indicators) of the harvest and *kaitiaki* practices related to Tītī in the Hauraki community had become increasingly diluted over the past 30 years. Reasons suggested for the erosion of this knowledge included: (1) the perturbation to traditional Māori life brought about by colonisation over the past 100–150 years; (2) the relatively recent movement of Māori away from their traditional lands and resources into an urban environment; (3) a general decline in the practice of harvesting Tītī; and (4) that the knowledge was possessed by only a few families within the *iwi* over the past 50 years. Interviewees felt that the latter reason had made the mātauranga vulnerable to loss. Even so, one asserted that certain elders within the Hauraki iwi still possessed a robust understanding of mātauranga related to the Tītī, although acknowledged that few of these individuals remained. Most interviewees agreed that to reverse the loss of mātauranga, the knowledge had to be "lived once again". They urged that the Tītī harvest become an annual activity for the iwi and supported with new initiatives such as *hui* (gatherings) and elder–youth *wānanga* (learning groups).

> *In the earlier years, it was quite different. It [the harvesting of Tītī] was part of the lifestyle of our old people really. They lived their mātauranga. Our tūpuna [ancestors] used to go out to Ohinau (fig. 1) in October and November and stay there for weeks while they harvested the Tītī and planted the kumara crops. Then they would go again later after the first frosts had occurred on the mainland and harvest their kumara and another species of muttonbird [species not identified].* (R.B. 2006)

## Changes in Tītī abundance

Birders identified a range of harvest-based indicators that they used to monitor Tītī population abundance. One interviewee whose family had historically harvested Tītī indicated that he and his father observed an initial decline in chick abundance on the Ruamaahua Islands during the late 1980s, with the scale of decline becoming very noticeable by 2003. The interviewee recounted his father making the comment in the late 1980s:

> *The birds don't seem to be here. They don't seem to be coming back to the islands like what they use too.* (V.T. 2007)

This particular interviewee felt that the harvest would never go back to the same level as it was 30 years ago because of the human and environmental pressures now facing the birds. He also reported that prior to 1990 it was usual for a birder to harvest two chicks from each burrow entrance. This was supported by interviewees from another family who referred to burrows in the past as "always being full" [of chicks]. However, they had also observed a decline in burrow occupancy during the early 1990s. Another interviewee reported that the number of burrow entrances on the Ruamaahua Islands has remained the same, but now you attempt about three holes and only get one chick.

> *In the past, when you got to the Aldermens [Ruamaahua Islands], it was just one mass of holes. You could sit in one place and stick your hand in. But you don't stick your hand in too far at the start because the bird up this end [of the burrow] will peck at it. You go in and pull it out, then another will be about a foot away, and another a bit further on until you clean it [the burrow] out. Then you move a bit. The softer the ground the deeper they [the birds] go. And you'll know if you get three or four birds from one entrance, then you know there are three or four different tunnels underneath.* (B.C. 2007)

The oldest interviewee recounted parties of 100–150 birders going out to the islands to harvest during the 1930s and 1940s and returning with around 15 000 birds. However, three interviewees reported that the size of the birding parties had shrunk to 10–20 birders by the late 1980s. Prior to this it was common for a birding party of 7–10 individuals to harvest about 1000–1200 chicks over a day (4–5 hours of birding). It was even possible for a single birder to harvest 250–300 birds in a day off the Ruamaahua Islands, but after 1990 an individual tally declined to about 50 chicks per day. Similarly, another interviewee who had birded on nearby Mahurangi (Goat) Island (fig. 1) recounted how in the early 1970s his two uncles and father could easily catch up to 120 chicks between them in the first hour of harvest. In contrast, records from 2005 and 2006 show that parties of between six and 10 birders harvested a combined tally of less than 100 chicks, although the majority of these individuals had little practical harvesting experience. The older birders linked the erosion of mātauranga indirectly to declines in the Tītī population. It was recognised that the optimal environment for passing this knowledge on between generations was during harvesting expeditions out to the islands. However, with the decline in bird abundance fewer people were harvesting and as a result knowledge was not being passed on. Unfortunately, in more recent times this knowledge loss has been exacerbated by the passing of the more experienced birders.

## Variation in burrow density and chick quality

Birders were aware of the patchy distribution of Tītī burrows on some islands. They reported that Tītī burrows were most dense around the peripheries of inshore islands (e.g., Slipper and Rabbit islands; fig. 1), whereas burrow density was higher and more evenly distributed across the landscape on the Ruamaahua Islands. They attributed this difference to the greater abundance of birds and forest coverage on the Ruamaahua Islands. On Slipper and Mahurangi islands, where areas of forest had been removed for farming, burrow density was much lower in the grass areas, especially where ungulates such as goats, *Capra hircus* Linnaeus, 1758, were present. Birders were acutely aware of these differences because of the manner that they had to move around the islands. It was reported that on inshore islands, an individual could stand up and walk between burrows, whereas on the Ruamaahua Islands they were required to tentatively crawl around to avoid collapsing burrows. One experienced birder recognised a greater abundance of small seabird species on inshore islands (e.g., Shoe, Rabbit and Penguin islands; fig. 1) compared with the Ruamaahua Islands. He postulated that interspecific competition for nesting space was the reason for this, and that the larger, more abundant Tītī could outcompete the smaller burrowing seabird species for breeding space on the Ruamaahua Islands.

Birders also recognised spatial differences in the quality of Tītī around some islands. One reported that his family used to target the burrows on the southeastern slope of Ruamahuaiti because the chicks in these burrows were generally larger than those from other parts of the island. Similarly, two birders from a different family explained that their father taught them that larger chicks were usually harvested on the eastern sides of the islands. The suggested reason was that the sun rises in the east and warms the burrows and eggs during the incubation phase, so the eggs in these burrows tend to hatch earlier.

Of the five most experienced birders interviewed, four had observed differences in chick growth between years. It was noted that in years of poor chick growth, for every 150 burrows prospected approximately six entrances would have dead chicks present. Interviewees attributed years of poor growth to lower food abundance or availability brought about by overfishing of Tītī prey species (e.g., squid) or climatic perturbations. However, none of the birders indicated that years of poor chick growth had increased in frequency.

## Hauraki resource management strategies

A range of traditional harvest management strategies used by Hauraki were identified by interviewees (table 2). One interviewee highlighted the importance of self-government and internal controls in caring for and managing the food grounds. She believed that Hauraki should determine who goes to the islands, and how they should conduct themselves while out there. The interviewee illustrated her point by explaining how her elders tended large vegetable gardens, and as children they were never permitted access to play for fear that they would damage the crops. Younger family members were tutored in how to tend and respect the gardens before gaining entry. She firmly believed that the same principles should apply to harvesting Tītī on the Ruamaahua Islands. Only once an individual has demonstrated the appropriate skill base and respect should they be granted access to the islands. Without this system of internal controls there will always be the very real potential of someone damaging and destroying the resource for the rest of iwi. Two interviewees reported that this happened when poachers accessed the islands to harvest Tītī. Without consideration for the tikanga, they were reported to have overharvested and indiscriminately collapsed breeding burrows.

The oldest interviewee described how his grandfather used to control the harvest and inform the people of the appropriate time to harvest the Tītī chicks each season. His grandfather would also determine the number of chicks that could be taken and the number of harvesters that should go out to the islands:

**TABLE 2**
**Concepts and traditional resource strategies used by Hauraki muttonbirders to manage Tītī populations**

| Concept | Traditional management strategies |
|---|---|
| Respect for species and its habitat | Teachings and directorship of harvest should come from *kaumātua* (respected elders).<br>You do not prepare or eat your food where you catch it. |
| Reducing the demographic impact | Vital life history stages (e.g., adults) are not harvested.<br>Harvest at the appropriate development stages. To protect your future breeding population do not harvest well-developed chicks.<br>Timing of harvest important to minimise disturbance, interference and desertion of adults. |
| Allowing for escapement | Harvest only occurs during a designated period.<br>Rotation or resting of islands harvested each season.<br>*Tohu* (environmental indicators) are used to determine whether harvest should proceed or not.<br>Chicks are only caught while down burrows and never at night when they emerge later in the breeding season to fledge. |
| Protection of habitat | Access to populations controlled or limited to specific *iwi* (tribe) or individuals within an iwi.<br>Digging should be minimised to avoid damage to burrows. |
| Enhancement of habitat | The digging of burrows can maximise and create breeding space which is advantageous to bird population. |
| Provision of refugia | *Tapu* (sacred rules) was used to restrict access or harvest to specific areas or islands. |
| Minimisation of waste | Do not harvest more of the resource than you can process effectively. |

*If there were birds everywhere, he would let the others [in the iwi] know and that's when they would take their young people out to teach them [how to harvest].* (B.C. 2007)

He also recounted how his grandfather would ensure that everyone received birds, including those families that could not go out to harvest. One experienced birder recognised that the timing of the harvest was important for maintaining the Tītī population. He maintained that chicks should not be harvested too early, even in seasons when they were well developed, because the adults will still be provisioning their young. He suggested that the adults will sense the premature loss of their chick and become discouraged from returning to the island to breed in the future. Therefore, the timing of the harvest, which begins on the first Saturday in November, is considered to be quite crucial.

All interviewees asserted that only the chicks were taken in the harvest. To kill adults was a serious transgression of tikanga. Two experienced birders explained that harvesting chicks at an inappropriate developmental stage could be detrimental to the population. They explained that underdeveloped Tītī chicks were poor eating and should be left to grow further, whereas chicks that were developmentally advanced, with more than half of their body covered by adult feathers, should not be harvested because they were the better-quality birds that are most likely to come back to the island and breed in the future. The interviewees felt it made sense to let these birds escape to ensure a future breeding population. The appropriate chick to harvest was one which still had down covering all of its body, but had adult plumage on its wings.

*You will notice it when you're muttonbirding. You will see that the adult feathers are starting to show up on the chicks, but they still can't fly. If you were to harvest the more advanced chicks, the old people would say "hey you brought the bloody adults back", and they would give you hell. Its true, you have got to leave some of the better chicks so that the population replenishes itself.* (B.C. 2007)

Three interviewees indicated it was disrespectful to the bird and a violation of traditional teachings to pluck, clean or eat the Tītī on the islands where they were harvested. They explained that these protocols were also commonly applied to other food resources such as Pipi, *Paphies australis* Gmelin, 1791, and Green-lipped Mussel, *Perna canaliculus* Gmelin, 1791. Shellfish was never opened or eaten while you were still working in the inter-tidal zone. Interviewees believed that cutting out the oil-filled crops of chicks and leaving them in the entrances of burrows or plucking and cleaning the chicks on the island would deter the adult birds from breeding there in the future. As one interviewee recounted:

*I remember my Aunty going off the deep end a couple of times because someone had gone out to the islands and harvested Tītī, then prepared the birds on the beach. That really annoyed her. All I can remember is her saying is that "they plucked the bloody things on the beach and cleaned them on the beach and just left the remains there. You don't eat kai [food] in the cupboard." She'd bring the whole bird back to her place and clean them all up at her place at Whiritoa.* (J.L. 2007)

Even so, another three interviewees reported cleaning and preparing the birds on the islands when they camped out overnight. All three interviewees described how the feathers or intestines of chicks were either buried on the island or disposed of in the ocean. No offal or feathers were left lying out around the birding grounds.

There was substantial division amongst interviewees regarding the collapse or digging of access holes through the roof of breeding burrows to facilitate the harvest of Tītī. Four interviewees felt strongly that collapse and the digging of burrows should be avoided at all costs because it would deter the adult birds from returning and using the burrow the following season. They believed if the burrow roof was damaged near or over the nest chamber, the burrow would leak and make it uninhabitable for the birds. One interviewee indicated that digging and repairing the burrows would be very time-consuming because of the friable substrate on the Ruamaahua Islands. In contrast, two birders described how shortening a burrow by digging at a point along its length was beneficial to the Tītī population because it increased the available breeding space on the island. They described how if a harvester could not reach a chick with their arm extended down a burrow (~ 1 m), then they would dig or punch through the burrow roof at the extent of their reach and reinsert their arm at the new opening to remove the chick. The soil from digging the hole was then used to create a back wall for the forward section of burrow, while the new opening was fashioned to form the entrance to the rear half of the burrow. The birders believed that this activity in effect created two burrows for the following season. They acknowledged that this technique only worked with burrows that were close to the surface.

Interviewees referred to the more recent conservation practice of reducing the exposure of Tītī to harvest by only having a short period of harvest on the islands (first two weekends in November). Minimising the time that chicks are exposed to harvest increases the likelihood that more birds will escape and return to breed in the future. The closure and rotation of harvest around different islands was another method used by Hauraki birders to rest populations and spread any potential impacts of harvest. One interviewee indicated that it was common practice for the Ruamaahua Islands to be closed and the inshore islands (e.g., Shoe, Slipper, Rabbit and Penguin islands – fig. 1) to be opened to harvest, or vice versa. However, he reported that in 1984, his mother had recommended to wildlife authorities that a rāhui be placed over the Ruamaahua Islands for a number of seasons because bird numbers had remained low. Two interviewees also indicated that Ruamaahuanui was rarely harvested because the island was harder to land on. They suggested this island acted as a partial refuge for the birds and a source of future breeders to the other islands in the group.

One interviewee referred to how *tohu* (signals or indicators) were likely to have been used to determine whether a harvest occurred or not in a year, or whether a harvest was large or small. The interviewee indicated that the old people had a particular time period for harvesting but that sometimes the weather was too rough and the birders could not get out to the islands. She believed her ancestors would have seen those conditions as a sign to stay home and not to go out to harvest, whereas today these links might not be made.

## DISCUSSION

### Re-establishing the relationship with Tītī

The harvest of Tītī has a key role within Hauraki tradition and identity. The ability to provide traditional foods reflects the mana of the iwi and their capacity to successfully manage their resources. It is considered by many Māori to be *whakama* (shameful) to receive visitors and not serve them a food sourced from the local area. This practice is fundamental in defining the iwi and/or individual as a kaitiaki for the resource. The Kererū, New Zealand Pigeon, *Hemiphaga novaeseelandiae novaeseelandiae* Gmelin, 1789, and Sooty Shearwater, *Puffinus griseus* (J.F. Gmelin, 1789), have similar cultural significance for the iwi of Tūhoe and Rakiura respectively (Taiepa *et al.* 1997, Lyver *et al.* 2008). The re-establishment of native bird harvests is seen by some Māori as the right to express their identity – a desire driven as much (if not more) by the cultural, social and spiritual significance of the practices associated with the harvests, as by the actual need for food (NZCA 1994, King 1994, Kirikiri & Nugent 1995). The health benefits of traditional foods have become widely recognised with increases in diseases like diabetes amongst Māori in New Zealand (Cambie & Ferguson 2003, Bovell-Benjamin 2007). Therefore, there has been an increasing emphasis to move away from the current westernised energy-dense diet and include a greater proportion of traditional foods (Murphy *et al.* 2003).

It was recognised that community isolation from the Tītī harvest over the past 20–30 years has eroded the mātauranga and left just a few older individuals with the more comprehensive knowledge base. With the decline in Tītī and isolation from the resource, the ability of Hauraki to maintain and build on traditional harvest practices has diminished. The current iwi authorities (Hauraki Māori Trust Board and Ruamaahua Islands Trust) have expressed the desire to become more actively involved in the restoration of Tītī populations using the best information available to them through science and their own mātauranga. The restoration of a sustainable Tītī harvest would reflect favourably on the iwi's abilities and mana as a kaitiaki. It would also demonstrate to the wider conservation community that Māori have the capacity to manage their own resources. However, iwi face the internal issue of how a future harvest might be regulated. In the past, individual influential elders experienced in the practice of birding controlled the harvest process and who participated in it, but in their absence Hauraki need to determine what management structures might be appropriate for the current situation.

The protection of some indigenous wildlife harvest rights through the post-colonial era has maintained the mechanism and opportunity for the continued development and transfer of mātauranga within those cultures (Taiepa *et al.* 1997, Beaton 1990, Ferguson & Messier 1997). Gilchrist *et al.* (2005) reported that quality of mātauranga was better for wildlife species with which local Inuit peoples had greater familiarity through harvest or year-round contact, or both. Similarly, the development and intergenerational transfer of Rākiura mātauranga continues through their participation in an annual harvest of Sooty Shearwater chicks on islands adjacent to Stewart Island. As one Rakiura informant stated:

> *In life there is muttonbirding, and then there is everything else.* (S. Bull pers. comm. 2005)

Even so, it is important to point out that indigenous peoples have extensive knowledge of species that they may not harvest and that all cultures are intellectually curious about the environment around them and that extends beyond direct connections to seemingly utilitarian matters like "knowing your food" (Nabhan 2000). Hauraki acknowledge that Rakiura have a healthy knowledge base that they can potentially learn and benefit from. However,

they also highlighted subtle differences in the way the harvest was conducted in the Hauraki region, and asserted how it is essential for them to preserve their own mātauranga and tikanga. Only then will the Tītī harvest, and the identity and mana that comes from this harvest, remain with Hauraki. It was also recognised that a combination of traditional and contemporary methods may need to be employed to ensure the persistence and communication of the knowledge within the iwi. Central to this process must be the transmission of knowledge across generations. New transmission techniques might include hui and wānanga, while re-establishing the harvest under the guidance of *tohunga* (experts) or kaumātua will ensure the persistence and growth of the mātauranga.

## Understanding changes in wildlife populations

Most current seabird harvesting goes scientifically unmeasured (Moller 2006). However, monitoring population baselines and trends in harvest rates and prey abundance can be an important component of mātauranga (Usher 2000, Kitson 2004, Moller *et al.* 2004, Gilchrist *et al.* 2005, Lyver *et al.* 2008). Through harvest-based indicators Hauraki birders were able to provide insights into the decline of the Tītī on the Ruamaahua Islands over a 50–60-year timeframe. The birders' understanding of breeding patterns was sensitive enough to detect that declines had occurred in chick occupancy, but not burrow density. This understanding is corroborated by burrow density surveys conducted in 1972 and 2006/07 (Fogarty & Douglas 1973, Jones *et al.* 2008), which show that, on average, burrow densities have not changed, and may have even increased slightly. In addition, preliminary evidence from recent burrow occupancy surveys suggests that chick occupancy is significantly lower on the Ruamaahua Islands than on other islands that have only Tītī present (e.g., Moutohora – Lyver *et al.* unpubl. data; Bethalls Beach – G. Taylor pers. comm. 2007).

In this case study, no scientific monitoring program had measured or detected the decline in Tītī populations on the Ruamaahua Islands. All evidence and reports of a decline had come from the Hauraki birders. Similar examples can be found in the Canadian Arctic where two cases of dramatic population declines (Common Eider, *Somateria mollissima sedentaria* (Linnaeus, 1758); Ivory Gull, *Pagophilia eburnea* (Phipps, 1774)) were brought to the attention of the federal wildlife management agency, the Canadian Wildlife Service, by local Inuit hunters (Gilchrist *et al.* 2005). Few, if any, scientific monitoring programs can provide population trend information over two to three generations for a long-lived seabird like the Tītī. At the very least, the Hauraki mātauranga can: (1) describe historical carrying capacity levels; (2) provide goals for restoration; or (3) be used as a prioritisation tool for focusing intensive research studies.

As low Tītī numbers persisted, Hauraki birders began to observe the effects through declining catch rates and lower annual tallies. To some degree Hauraki birders were able to buffer their harvest against initial declines in Tītī abundance by targeting islands, or areas on islands, of high burrow density. This scenario typifies the curvilinear relationship that often exists between catch-per-unit-effort (CPUE) and population abundance when a hunter is faced with a declining resource (Moller *et al.* 2004). By using mātauranga and technological advances to harvest at times and places of high prey concentration, the hunter can buffer the decline in their CPUE, even when the population is declining at a greater rate. Rakiura birders had a much greater opportunity to buffer the declines in their harvests of Sooty Shearwater chicks by harvesting in areas of high chick density and when chicks were aggregated above ground (Lyver *et al.* 1999, Lyver 2002). Even so, the rate of decline reported by Hauraki birders may be an underestimate of the true population decline. Birders recognised areas on islands (e.g., island edges and slopes with an eastern aspect) as having better quality and greater densities of chicks, and by targeting these areas buffered their harvests against decline. Surveys conducted on the Ruamaahua Islands in 1972 showed that burrow densities were most concentrated on gentle slopes under high canopy with little undergrowth (Fogarty & Douglas 1973).

Years in which Tītī chicks have experienced poor growth have not increased in frequency indicating that food supply in most years has not affected the provisioning of chicks and recruitment in later years. Rakiura birders also linked changes observed in harvest patterns to declines in titi abundance. Rakiura mātauranga predicted that years with larger and fatter chicks will have greater chick abundance; however, over the past 15 years birders reported years that chicks were fat, but there were fewer in the burrows than expected (Lyver 2002). This suggested to the birders that food was still available, and that some other factor(s) were causing declines. A similar scenario was faced by Hauraki birders.

Higher species diversity of smaller burrowing seabirds (e.g., Fluttering Shearwaters, *Puffinus gavia* (J.R. Forster, 1844); Little Shearwaters, *Puffinus assimilis haurakiensis* Gould, 1838; Diving Petrels, *Pelecanoides urinatrix urinatrix* (J.F. Gmelin, 1789); White-faced Storm-Petrels, *Pelagodroma marina* Latham, 1790) on some inshore islands (e.g., Penguin Island, Rabbit Island) compared with the Ruamaahua Islands was attributed to fewer Tītī present on these islands. However, the lower chick occupancy on the Ruamaahua Islands detected in recent surveys suggests that the historical harvest of Tītī chicks from the islands may have in some way benefited these other burrowing seabird species that breed later in the year (e.g., freeing up of breeding space), and potentially contributed to a change in species composition on the islands. If species composition has changed then any recovery of the Tītī population may not be straightforward. The rapid expansion of the Buller's Shearwater, *Puffinus bulleri* Salvin, 1888, (which are ~100 g smaller than Tītī) population on Aorangi, Poor Knights Islands, after Wild Pig, *Sus scrofa* Linnaeus, 1758, eradication (S. Bartle pers. obs. 1968) was at the expense of other petrels, including Tītī and Fluttering Shearwaters (Harper 1983).

## *Mātauranga* limitations to detect changes in meta-populations of seabirds

Traditional methods used by Hauraki to detect changes in the Tītī meta-population on the Ruamaahua Islands were largely harvest-based. Unlike in earlier years (pre-1900s), when people would stay for extended periods (4–6 weeks at a time) on some birding islands, recent contact with the Tītī population has been limited to 2–4 days per year and is mainly restricted to daytime observations. This has meant that impressions and hypothesis generation formed about the resource by Hauraki birders have come from only a small window in the birds' life history. The nocturnal and subterranean breeding habits and the large at-sea component of the Tītī's life history also

compound these efforts. In contrast, some Rakiura birders can spend up to 10–11 weeks living on the birding islands and use a wider suite of indicators – e.g., (1) numbers of adults returning to the islands at night; (2) smell of the islands; (3) level of noise of adults returning to and leaving the island each day; (4) amount of trampling caused by the birds along tracks; (5) catch rates; (6) condition of chicks; (7) emergence time of fledglings; and (8) changes in overall burrow densities around the islands – to evaluate the status of the Sooty Shearwater population (Lyver *et al.* 1999, Newman & Moller 2005, H. Moller pers. comm. 2008).

By using only harvest-based information to monitor populations it is difficult to interpret anything about non-harvested (control) populations. It could be possible that Tītī colonies on non-harvested islands are not in decline but Hauraki would have no way of knowing this without using non-harvest-based sampling techniques. A similar scenario confronted Rakiura birders. Declines in the Sooty Shearwater population over the past 15 years were detected by Rakiura through their harvests (Lyver *et al.* 1999); however, studies of beach patrol records and a non-harvested population on the Snares Islands, located 200 km south of the Rakiura Tītī Islands, corroborated these observations (Scofield & Christie 2002, Miskelly *et al.* 2001). Evidence pointed to a climate-based mechanism that could be potentially impacting adult Sooty Shearwater survival (Lyver *et al.* 1999, Bragg *et al.* 2007).

It is very difficult to differentiate between harvested- and non-harvested-related factors impacting a seabird population. It would be highly likely that there are feedback loops related to harvesting behaviour and the natural demographic patterns of Tītī (Newman & Moller 2005). Interviewees postulated that removing chicks from breeding burrows at the wrong time could increase the risk of colony abandonment by the adult birds. If harvest is in some way linked to Tītī ecology, then changes to harvest rates will affect the bird's ecology making it very difficult to disentangle the process. In addition, differential movement rates between harvested and non-harvested meta-populations may remove evidence of harvest effects.

## Resource benefits of Tītī harvest strategies

Interviewees acknowledged that absolute protection of adult Tītī was paramount under Hauraki customary lore and regulation. This included not harvesting adults or disturbing adults in any way through the breeding season. This concept is widely supported by many seabird studies which have shown that small changes in adult survival have a large impact on population growth rates (λ) (Hamilton & Moller 1995, Hunter *et al.* 2000). Also, a long-term Tasmanian study of Short-tailed Shearwaters, *Puffinus tenuirostris* (Temminck, 1835), has demonstrated that repeated researcher disturbance of the colony contributed to the decline of the colony, increased burrow desertion, and decreased breeding success and recruitment of young adults (Serventy & Curry 1984). Selective harvesting of chicks at certain stages of development could also reduce harvest impact. The Hauraki practice of leaving the more advanced birds to fledge indicates their understanding of the relationship between chick quality and recruitment. A similar strategy is also employed by "Nessmen" that harvest 2000 Northern Gannet, *Morus bassanus* (Linnaeus, 1758), chicks from Sula Sgeir, an islet some 90 km north of the Isle of Lewis in the Outer Hebrides (Beatty 1992). Younger downy gannet chicks and the slimmer, developmentally more advanced individuals (with mostly adult plumage) are left to fledge, while chicks in the intermediate stage of development are harvested. In contrast, Rakiura muttonbirders prefer larger and more developed chicks, which have a higher survival and probability of recruitment (Sagar & Horning 1997, Hunter *et al.* 2000). However, the contribution of chick survival to λ is small so harvest selectivity may have little demographic impact unless the harvest is very intense or it is coupled with elevated adult mortality (Hunter & Caswell 2005). It is possible that the declines in the Tītī population on the Ruamaahua Islands could be largely driven by extrinsic factors impacting crucial demographic phases (e.g., adult survival) away from the islands. If survival rates of adult Tītī have decreased over the past 50 years, the removal of chicks at historical levels could be forcing the harvest into a state of unsustainability, and reducing the effectiveness of traditional breeding-based conservation strategies.

Harvesting too early or leaving the remains of chicks on the birding grounds were deemed to be serious transgressions of Hauraki harvesting tikanga because they were believed to lead to burrow abandonment and higher rates of emigration by adults from the islands. Moller (2006) also postulated that disturbance of breeding by harvesting could exacerbate impacts of simple additive mortality by promoting increased emigration in pre-breeders. This could quite easily apply to the breeding population as well. Violation of customary lore or a display of disrespect towards a species is commonly used by indigenous cultures to explain declines or disappearances in wildlife. Tūhoe perceived that the recent practice of plucking kererū in the forest was partly cause for declines in its population because leaving the bird's feathers scattered around the forest signalled disrespect. This value-belief ethos is also evident in other indigenous cultures, such the Siberian Yupik, where the belief that an animal (especially a sea mammal) killed, butchered, consumed and disposed of according to specific respectful practices would ensure the animal would follow the eternal cycle of revival and availability to the people (Krupnik & Vakhtin 1997).

The protection of burrows from collapse and digging by some was perceived to be a critical component of managing the Tītī population. Reduced productivity at seabird colonies caused by humans collapsing burrows and compacting substrata is a recognised threat on the Rakiura Tītī Islands (Moller 2006). The reasoning behind splitting existing burrows in two is that it creates additional breeding space and releases the population in part from density effects (e.g., interference competition) leading to increased egg, chick and juvenile survival, and a higher carrying capacity on the islands. Alternatively, if done wrongly it could damage burrow structure and have quite the opposite effect. If inexperienced birders were going out to the islands to harvest unsupervised, they could cause irreparable damage by either harvesting chicks in the wrong developmental stages or collapsing burrows indiscriminately. This was supported by a number of interviewees who felt that the core part of Hauraki's management strategy for the islands needed to include a rigorous tuition process for all new birders going out to the islands. A study with Rakiura birders found that 94% (n = 17) of interviewees that had harvested chicks from burrows referred to the protection of burrows as a "very important form" of their traditional management (J. Kitson unpubl. data). Rakiura tikanga dictates that if a hole is dug over a nest chamber or burrow passage to reach a chick, it must be plugged. However, the plugging

of burrows occurs on some islands only once burrows have been shortened to maximise breeding space (H. Moller pers. comm. 2007).

Rāhui is one of the core attendant management practices used by Māori to serve the environmental ethic of "conservation for human use" and ensure resource sustainability (Roberts *et al.* 1995). Hauraki utilised a form of rāhui by adopting a harvest rotation policy whereby Tītī meta-populations on islands were "rested" from harvest every second year or so to reduce the impacts of harvest (e.g., adult disturbance, chick removal). A similar strategy is used by Cree stewards who use their knowledge of changes in beaver numbers, size of colonies, size of litters and frequency of abandoned or new colonies to direct the intensiveness of hunting within their territories (Feit 2004). Partial refuge islands (e.g., Ruamaahuanui), where harvest occurred infrequently, were also used by Hauraki to provide recruits to support harvested populations on the nearby islands (Hongiora and Ruamaahuaiti). Island refuges is a relatively common concept that has been implemented in the Seychelles, Indian Ocean, to safeguard a Sooty Tern, *Onychoprion fuscata* (Linnaeus, 1766), population by providing recruits for the neighbouring harvested colonies of Desnoeufs and Îlot Fregate (Feare 1984). The resource management strategies presented provide evidence that Hauraki harvesters were able to synthesise and interpret their diachronic observations in a population context. In doing so, it also opened up the possibility of understanding how these demographic components could be influenced by environmental parameters across a range of temporal and spatial scales.

## CONCLUSION

Our study illustrates the way Hauraki mātauranga can understand and hypothesise about complex island ecosystem effects that have important management implications. Mātauranga, by its holistic nature, provides multi-species and community-level information even where the primary focus is on a key species. Hauraki mātauranga has proven to be a useful tool for highlighting trends in the Tītī population and management options that promote (1) the provision of refuge; (2) habitat protection; (3) spatial spreading of impact; (4) escapement,;(5) minimising of disturbance; and (6) harvesting of life-stages that have minimal demographic impact. Addressing the relationships between these strategies and demographic outcomes, and their potential benefit in the current environment, can now be the focus for future research and analysis. Consideration of the demographic effect that each strategy or combination of strategies has had on the Tītī population can also guide decision making about harvest management on the Ruamaahua Islands.

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# LOOKING AFTER YOUR GROUND: RESOURCE MANAGEMENT PRACTICE BY RAKIURA MAORI TITI HARVESTERS

by Jane Catherine Kitson and Henrik Moller

(with one text-figure)

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The annual harvest of Titi, *Puffinus griseus*, on islands adjacent to Rakiura (Stewart Island) by Rakiura Maori is one of the last large-scale customary uses of native wildlife in New Zealand. This study investigates whether Rakiura Maori harvesting practices constitute common property resource management and how these practices relate to the sustainability of Titi harvests. Semi-directive interviews were conducted with 20 experienced Titi harvesters and elders to record their matauranga Maori (Traditional Ecological Knowledge) and tikanga (lore) that govern this harvest. Access to the resource is shared and controlled by birthright. Informal and formal sanctions enforce the rules that promote sustainable use by protecting island habitat and adult birds, and minimising disturbance. These rules and other aspects of Titi harvesting practice conform to common property resource management theory. While adhering to the main resource rules, localised flexibility in management provides harvesters with the ability to adapt to changing environmental and social circumstances. Adaptive co-management of Titi harvesting has included turning lore into law to reinforce social institutions, integration of science and matauranga for sustainability assessments, and application of new ecosystem management tools. Environmentality is promoted by local self-government of Rakiura Maori Titi harvesting practice and island ecosystem management.

**Key Words: *Puffinus griseus*, Titi harvesting, Traditional Ecological Knowledge, lore, adaptive co-management, environmentality.**

## INTRODUCTION

The term Traditional Ecological Knowledge (TEK) refers to "a cumulative body of knowledge, practice, and belief, evolving by adaptive processes and handed down through generations by cultural transmission, about the relationship of living beings (including humans) with one another and the environment" (Berkes 2008: 7). It is an attribute of societies with a long-term continuity of resource use at a particular place, especially indigenous peoples' customary use of wildlife and natural resources. TEK and associated lore is potentially crucial in determining whether customary harvests of wild foods are sustainable in a modern world of globalisation, commercialisation of markets, new harvest technologies, rising population pressure and invasive species (Newman & Moller 2005).

Sustainable customary use of wildlife by indigenous people often involves an added challenge because the resource may be owned by an entire tribal group or harvested from communally owned land. Common property theory applies to these situations where (i) the exclusion of potential users is difficult, and (ii) the activity of each user subtracts from the welfare of all the others (Berkes 1989, Feeny *et al.* 1990). Achieving sustainable common property resource management requires the group to control access of all potential users and to have common property institutions. The latter include decision-making arrangements where rules for resource harvesting and sharing are made and enforced among the users themselves (Berkes 1986, 1989). Once access and resource use rules are in place, both the costs and benefits of any management action will be borne by the same individual or group, thus providing an incentive to conserve. The alternative has been characterised by Hardin (1968) as "the tragedy of the commons".

Resource use rules can be complex, vary between communities and be fine-tuned to spatial and temporal variation in ecological conditions. Their implementation may involve an intimate belief system that incorporates religious beliefs, cultural rituals and other social mechanisms alongside environmental and natural history observations (Gadgil *et al.* 1993, Colding & Folke 1997, 2001, Berkes *et al.* 2000, 2003, Berkes 2008). Inclusion of the word "traditional" does not imply that TEK is static – indeed it builds from experience and the incorporation of useful aspects of modern technological innovation (Gadgil *et al.* 1993, Stevenson 1996, Lyver & Moller 1999a, b, Lyver 2002, Berkes 2008). Learning by doing is fundamentally important for "adaptive co-management"; the way indigenous communities respond to socio-economic and environmental changes and challenges to local governance and management (Berkes & Turner 2006, Turner & Berkes 2006).

Māori, New Zealand's indigenous people, use the term *mātauranga* for their knowledge and *tikanga* for their customary lore concerning resource management rules. Together they have guided natural resource use for centuries in Aotearoa New Zealand (Roberts *et al.* 1995, Kawharu 2002). This paper describes in detail how Rakiura Māori manage one of the last large-scale customary uses of native wildlife in New Zealand — their annual harvest of Tītī, *Puffinus griseus* (J.F. Gmelin, 1789), Sooty Shearwaters or muttonbirds. Rakiura Māori is a legally-defined term based on usufructuary rights to harvesting Tītī, as determined by *whakapapa* (genealogical links) to the harvesters present when the 1912 regulations were promulgated (Wilson 1979, Stevens 2006).

Some people, especially *Pākehā* (non-Māori people), dispute that Māori can adequately manage the New Zealand environment (Moller 1996, Taiepa *et al.* 1997, Newman & Moller 2005). This belief is seen in their submissions to the Southland Conservation Board on the proposed return of the ownership of the Tītī Islands to Ngāi Tahu ownership (Southland Conservation Board 1994) and their submissions

to the New Zealand Conservation Board on Customary Use of Wildlife (New Zealand Conservation Authority 1997). Those submissions challenged whether adequate TEK still exists amongst Māori; whether it can ever be sufficient to ensure sustainable use; and whether it can guide within new ecological conditions prevailing in New Zealand, or where new technologies may have made current harvest rates unsustainable. There were then no detailed case studies of Māori TEK applications for sustainable environmental management to evaluate these assertions, yet several New Zealand statutes directed government agents to apply Treaty of Waitangi and *kaitiakitanga* (Māori environmental guardianship) principles in their policy and operations (Moller *et al.* 2000).

Rakiura Māori established the overall Tītī research program in 1994 primarily to examine the sustainability of the harvest to ensure that the birds remain plentiful for their *mokopuna* (grandchildren) (Moller 1996, Taiepa *et al.* 1997). Research was in part motivated by decline in Tītī abundance (Scott *et al.* 2008), even though the absolute abundance is extremely high (Newman *et al.* 2008 a, b). The *kaitiaki* (environmental guardians) called the program *Kia Mau Te Tītī Mo Ake Tōnu Atu*, which means "Keep the Tītī Forever". An immediate priority of the program was to record the mātauranga of the Rakiura Tītī harvesters and the ways it might promote sustainability. We also sought to provide a detailed case-study of kaitiakitanga, to provide clarification of its nature and extent, and to aid government agents in the application of environmental statutes in Aotearoa. Additionally we sought to place a Māori example of TEK and environmental lore into the context of practices of Indigenous Peoples observed elsewhere in the world. Accordingly, the objectives of this paper are to (a) describe the mātauranga and tikanga that guide Tītī harvest management; (b) compare these described characteristics with the TEK of Indigenous Peoples elsewhere; (c) assess whether Rakiura harvesting practices constitute common property resource management; and (d) consider ways that mātauranga and associated tikanga might promote sustainability of customary harvests.

## RAKIURA TĪTĪ HARVESTS: AN ICONIC CUSTOMARY WILDLIFE HARVEST

Tītī is probably the most ecologically important seabird in New Zealand (Warham & Wilson 1982). The centre of breeding abundance is in Foveaux Strait and on 36 Tītī ("Muttonbird") Islands around the island of Rakiura (Stewart Island) (fig. 1). Tītī nest in burrows dug by the adults in deep and soft peaty soils under a low forest canopy consisting mainly of *Tūpare*, *Olearia colensoi* Hook., and *Tētēaweka*, *Olearia oporina* Forst., tree daisies.

The Tītī harvest is a defining cultural activity for Rakiura Māori that generates social cohesion and group identity (Waitangi Tribunal 1991, Stevens 2006). Historically, Rakiura Māori would have followed a resource gathering (*mahinga kai*) cycle. Their movements to different locations were according to the seasons; they followed the life-cycles of the animals and plants that they depended on for survival and cultural continuation (Dacker 1990). Outside the Tītī harvesting season, they would have been busy gathering other types of materials in different locations, leaving little opportunity to visit the islands. The only other abundant food sources on the islands outside the season were seals and sea-lions, which would have been far more easily obtained from accessible mainland coasts (Hawke *et al.* 2003). Archaeological data show that Sooty Shearwaters were widely exploited as a food and trade source in prehistoric New Zealand (Dacker 1994, Hawke *et al.* 2003), but their present large-scale use may have been a protohistoric phenomenon (Anderson 1997, but see Davis 1999 and Moller 1999 for a counter-view). "Muttonbirding" is a collective term for techniques and practices in which chicks of various Procellariidae seabirds are captured, processed and preserved for food (Anderson 1996, 1997). Traditional harvests are known from the north Atlantic, Australia, Raoul and Norfolk islands, and several places within New Zealand (Anderson 1996). Other than the Rakiura harvest, the only contemporary large-scale muttonbirding occurs in Tasmania, where Short-tailed Shearwater, *Puffinus tenuirostris* (Teminck, 1835), chicks are harvested commercially and recreationally (Skira & Wapstra 1980, Skira *et al.* 1986, Skira 1990).

On 29 June 1864, the Deed of Cession of Rakiura was signed between Māori and the Crown. Together with subsequent legislation, the Deed acknowledged the hereditary rights of Māori to harvest Tītī. "Beneficial Islands" were set aside for the use of Rakiura Māori descended from chiefs who signed the Deed. "Crown Islands" were mistakenly retained by the Crown for the use of other Rakiura Māori who wrongly missed out on allocations on the Beneficial Islands. Rights to harvest Tītī chicks each "birding season" (1 April–31 May) from 36 Muttonbird (Tītī) islands were first legally instituted in 1912. Associated harvesting regulations were amended in 1922, 1949 and 1978 (Department of Lands and Survey 1978, Wilson 1979). In 1998, the ownership of the Crown Tītī Islands was vested in Ngāi Tahu by the Ngāi Tahu Settlement Act (1998) and these islands were renamed the "Rakiura Tītī Islands".

The main periods of harvest are *nanao* and *rama*. During the nanao, which occurs from 1 April until mid/late April, the chicks are extracted from the burrows during daylight. Historically between nanao and rama there was also a *pahure* harvest phase; however, this is less commonly undertaken today. During pahure, birders pass over the birding ground during daylight for a second time, this time targeting "tarred" burrow entrances (those marked with down). The down is deposited by early emerging chicks that have come out at night to flex their wings and lose down, so tarring signals that a chick occupies the burrow. During the rama (or "torching" period) from late April until mid-May, the chicks are captured when they emerge from their burrows at night (Lyver 2000a, b, Hunter *et al.* 2000b).

Introduced rats (Kiore, *Rattus exulans* (Peale,1848); Ship or Black Rat, *R. rattus* (Linnaeus, 1758); and Norway Rat, *R. norvegicus* (Berkenhout, 1769)) are present on some Tītī Islands. Stewart Island Weka (*Gallirallus australis scotti* (Ogilvie-Grant, 1905)), an endemic rail, was introduced to the Tītī Islands early in the twentieth century by the Tītī harvesters for food. Rats and especially Weka depredate Tītī eggs and chicks and threaten harvest sustainability (Moller *et al.* 2003, Harper 2006, Dillingham *et al.* 2007, Harper 2007, Newman *et al.* 2008a).

## RESEARCH METHODS AND FRAMEWORK

Establishment of the *Kia Mau Te Tītī Mo Ake Tōnu Atu* research program involved prolonged and repeated discussion between all members of the Rakiura Māori community and researchers over two years before the actual work began (Moller

2001a). A "cultural safety" agreement was established between Rakiura Māori and the University of Otago research team to secure trust and protect intellectual property rights from the mātauranga and to ensure that research methodology did not violate Māori tikanga (Moller 1996, Taiepa *et al.* 1997). This agreement ensures that while scientific data from the program are jointly owned, the TEK remains the intellectual property of Rakiura Māori. The agreement also stipulates that all written work resulting from the research program is reviewed and commented on by the "Rakiura Tītī Island Administering Body", which consists of representatives elected by the Rakiura Māori community. Before submission, the

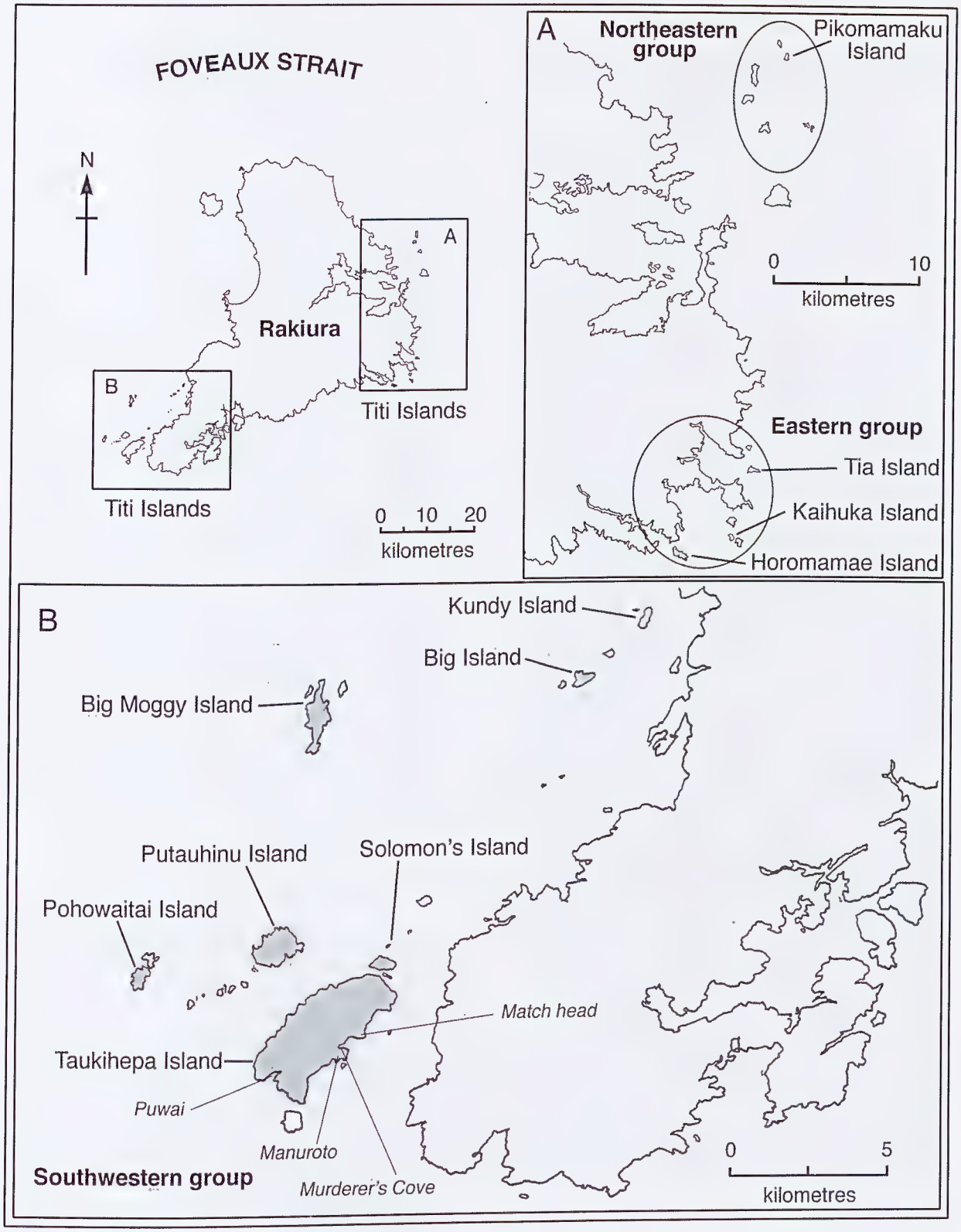


*FIG. 1 — Rakiura (Stewart Island) and adjacent Tītī Islands. Interviewees' birding islands are shaded and named. Specific birding areas (manu) are also named for Taukihepa.*

Administering Body validated this paper. There was much community concern about opening the traditional harvest to scrutiny of "outsiders" (Davis 2001, and discussion to papers in Howard & Moller 2001). Thus it was important to proceed slowly and build confidence amongst all participants that the researchers would treat the information shared with respect and represent it accurately in publications. Equally, the cultural safety agreement protected the researchers' scientific ethics by stipulating that the Rakiura Māori community could not prevent publication, irrespective of what the research concluded about the sustainability or otherwise of the harvest (Moller 1996, Moller *et al.* 2000). Repeated presentations and community peer reviews took place at annual *hui* (meetings) on *marae* (traditional Māori meeting places).

Although field-based ecological studies began in 1994, it was not until 1997 that trust was established sufficiently to begin interviews about mātauranga, the most culturally sensitive information of all. We interviewed 20 experienced muttonbirders, elders and *kaumātua* (respected elder, Garven *et al.* 1997). The selection of participants was non-random because we targeted older people who are more likely to have retained traditional management strategies. In many indigenous communities, elders play a key harvest management role – they are often the keepers, transmitters and interpreters of TEK (Berkes 2008, Berkes & Folke 2002). For the purposes of our project, the interviewees had to be over 50 years old, still actively muttonbirding, and living in the Southland area. This project had an additional purpose, in fulfilling a request that the oral history from the kaumātua of the Rakiura Māori community be recorded before the information was lost. Because of this, two kaumātua were interviewed even though they had not been birding since they were teenagers.

We used the standard practice and techniques employed by social scientists for identifying relevant informants (key informants, snowball techniques, triangulation, etc.) and conducting interviews (informed consent, open-ended questions; Denzin & Lincoln 2003). The Rakiura Tītī Islands Administering Body introduced us to two potential interviewees, who in turn suggested other elders to approach. The sample of interviewees consisted of seven females and 13 males who ranged from 55 to 85 years old (mean = 69 years). A married couple were interviewed together at their request. The sample represented approximately 30% of the possible candidates who fitted the selection criteria at that time. The heterogeneity of practices and traditions on different islands was addressed by interviewing representatives of various families from 11 different islands and 17 *manu* (family birding territories on Tītī breeding ground). The majority of interviewees had birded on the southwestern Tītī Islands (fig. 1), where birding is concentrated. However, other Tītī Islands were also well-represented by interviewees from an excellent cross-section of birding locations, shown in figure 1.

Four additional people approached for interviews declined. One refused because she apparently distrusted science and another distrusted researchers because a previous and unethical researcher had taped their conversation without her consent. The relatively high proportion of older birders interviewed and the low level of non-participation suggests that our information will be an unbiased representation of the views and knowledge of the older Tītī harvesters in the community.

All interviews were conducted by JK. Before the "official" interview took place, she visited the prospective interviewees, in some cases a number of times, and spent time building up trust. In this pre-interview period, she asked them questions of a general and personal nature; however, during the majority of this period the questions came from the prospective interviewee. As JK is of Rakiura Māori descent, it was important for the interviewees to find out how she related back to them, and to share accounts of her family members.

The interviews were unstructured (Lofland & Lofland 1995) or lightly semi-directed (Huntington 1998, 2000), where direction and scope of the interview mainly followed the interviewee's train of thought and any guidance to emerging concepts by the interviewee only occurred for clarification, or occasionally at the end of the interview, if the birder had not already covered the ground. There was no fixed questionnaire, though an interview guide was referred to for prompting further discussion if there was a lull in conversation. The interviews took place between November 1997 and early March 2000 and were audio-taped and transcribed. All interviewees received an audio copy and transcript of the interview. If the interviewee was willing, a copy of the interview/s was lodged at the Southland Museum and Art Gallery, sometimes with restrictions placed on access and future use. Seventeen interviewees granted permission for their interviews to be lodged at the archive. Although many of the interviewees have been going to their island since they were very young children, they were very concerned about being singled out as "authorities" on muttonbirding. Confidentiality was assured to them in any publication; hence this manuscript refers to the participants as Birder A, B, C, etc.

Spoken language can appear quite ungainly when typed on a page. To aid readability, quotations from the transcripts have been edited to remove fillers (such as "um", "ah" and "you know" etc.), repeated phrases, false starts, pauses, laughs, and the occasional grammatical error. Occasionally, changes were also made to ensure confidentiality. Square brackets indicate where we have added text to aid understanding. Three reviewers (two from our research team and an independent expert on ethnographic research) compared the original transcript quotations with the edited quotes in this text to ensure that any changes made were to aid understanding and keep the original meaning intact.

## RESULTS

### Access to the birding islands

Access to the birding islands is conferred through Rakiura Māori hereditary rights and is controlled in two ways, depending whether the island is a Rakiura Tītī Island or a Beneficial Tītī Island.

Any "Rakiura Māori" (as defined by statutory regulation) can apply for an annual permit for any of the Rakiura Tītī Islands. Applications for the permit are submitted to the Rakiura Tītī Island Administering Body. Before the Tītī season begins, permits for Rakiura Māori and their spouses are read out at an annual gathering of the birding community ("Permit Day"). Should a dispute arise from a permit, the individuals concerned have to prove their whakapapa in front of the gathering.

On Beneficial Islands, prospective birders must prove that they descend from one of the original owners of that island or, in the case of Taukihepa (fig. 1), the specific manu where

they propose to harvest. To become a "beneficiary owner", birders must inherit their beneficiary rights from their parent when they die. The younger generations in those families are called "potential beneficiaries" and are allowed to bird with the permission of the family member who holds the beneficial right. Historically, decisions on particular islands or manu are made by the beneficiary owners:

> *My old people, when they were going to the islands, we had no say whatsoever. You did as you were told, and it worked out all right. We weren't the beneficiaries at that stage, they were the beneficiaries, [and] we were only potential [beneficiaries]. Whatever my Dad said went, that was it.* (Birder E)

### Territoriality within islands

Territoriality often forms part of the control of access within common property management systems (Berkes 1986). The forms of territory systems found on the Tītī Islands can be broadly described as "closed" and "open". In a closed system, an area is allocated to each adult birder or each family on the island. In an open system, all the birders on the island can harvest anywhere on the island. The type of system in place and variations on these depend on the how the birders "worked" (organised the allocation of the resource) on that particular island, as the following quote from Birder C highlights:

> *Now we have our main manu [birding area] that is ours, and then if there's any left over, you share it per family. That's the way we work ours. The working rule[s] on all islands are different. I mean we work ours different from say likes of Pohowaitai. They work strips nanaoing but it's an open island for torching. But it all depends on the island. [The] likes of Putauhinu, they have their family manus regardless of how many on each manu. The people on the island make their own working rules.* (Birder C)

On some islands the type of territory system in place varies according to the development stage of the Tītī chicks. During nanao, some islands work in "strips", where all the birders work side-by-side, harvesting from one boundary to another on a pre-determined width of manu. For individuals the location of these strips could vary from year to year, or remain constant. During pahure, during which birders can go over the ground again before the rama period, access is generally open. Once rama begins, the islands become either open or closed systems. In those islands with open manu during rama, allocation management of territories occurs prior to the start of the season and is determined by the birders from that island. In some open systems, families allocate different times of night to harvest during rama, so that they do not deplete the number of emerged chicks available for capture. Some birders harvest during the early part of the night and others just before dawn, or they can take turns to harvest (Birder H; Moller 2002).

Historically, the kaumātua of the island determined birding territories. Boundaries are commonly demarked by tracks, creeks or other natural markers like edges of forest patches. Separate areas are allocated for men and women to work during the nanao on Horomaemae. Historically, the same happened on Big Island. On Pohowaitai, the allocation of harvest areas to groups of birders to work in strips during nanao is determined by one member from each group drawing a playing card; the person who draws the highest card has the first choice of area. Working strips requires that all the birders work together. If a day is to be taken off during nanao, then all the birders need to take that day off.

The territory systems are dynamic and systems on islands have changed over time. Some islands that had closed systems now have open ones, while Big Island has changed from a closed to open, and then back to a closed system over a lifetime. Birders have observed changes in the numbers of Tītī as natural successional changes in vegetation make the habitat more (or less) suitable for the birds. On Big Island this resulted in an increase in the number of birders wanting to work the area as numbers of Tītī increased. The island's closed system was opened as there was not sufficient land area to allocate individual manu, and the open system enabled the increased resource to be shared among a larger number of birders. As vegetation patterns continued to change, Tītī numbers declined to a level where fewer birders wanted to work the area, once again allowing the allocation of individual manu and a return to a closed system.

## TIKANGA: THE RULES

Certain traditional rules or *tikanga* apply to the harvest of Tītī. Beginning with the framework set out in Folke *et al.* (1998), we explored "rules" or the ways in which birders "look after" the manu/island, within several categories: protection of habitat, keeping the manu tidy, protection of breeding burrows, *tapu* (access limits), temporal restrictions of harvest, protection of adults, and waste minimisation. We then consider overarching concepts of sanctions, compliance, reciprocity and learning for sustainability.

### Protection of habitat

The interviewees placed a strong emphasis on rules for protection of habitat. Discouragement of unnecessary felling of live wood was strongly emphasised in nearly three-quarters of the interviews. In the past, firewood was a sought-after resource because of the reliance on open fires for use in the houses and for cleaning and cooking the chicks. Despite this high demand, the birders were only allowed to collect dead wood for firewood. Collecting firewood was something that the majority of the interviewees participated in as a child. Birder C recalls branding the dead trees in order to claim them as firewood, because there was such a demand for wood:

> *... through the season you didn't take trees that were standing and with all the houses on the island [with] open fires you'd have to go a long way [to collect wood], but [when] everybody would land [on the island] they'd rush and put their brand on the fallen trees that were closest, so if you were there early you got the closest trees.* (Birder C)

Fire was controlled to minimise the risk of the fire spreading, but also because historically, before the advent of modern communications, lighting a fire on the landings was the only way to signal that someone on the island was ill and needed evacuation:

> *You were not allowed to light fires indiscriminately anywhere because fires were used as a medium of communication from island to island. And fire[s] had to be lit in a certain place; because of the composition of the soil on the island being peaty it could burn on for years and years if it got into the ground.* (Birder D)

Fires are much less frequent in recent decades because coal and gas bottles for fuel can be transported by helicopter.

The cutting of tracks is necessary for access to manu, to

transport harvested chicks, aid navigation and demarcate manu. Tracks need to be clear enough for the birders' safety (i.e., avoid scratching themselves, poking an eye out, or hitting their head on a low hanging limb, etc.) but not too wide to cause unnecessary damage to vegetation. Birders try to use old established tracks rather than establishing new ones, so as to avoid damage to breeding burrows, as can be seen from this extract:

> *... you're taught from when you're old enough to use a slasher that you mustn't cut your track too wide and too far back. [It's] got to be clear so that you're not going to run into sticks or that in the night, but you're taught how to cut your tracks so that you're just trimming it back each year and you're not going [to] damage your trees. So that you're sort of weaving your way through the trees, more than opening up big tracks because you're relying on the roots of those trees to bind all those holes [Tītī burrows] together and keep everything intact. If a tree grows up in the way you'll have to sacrifice it because you shouldn't deviate your tracks. Because if you do, you're tramping a different part of the ground. What happens over the years is that the stuff that you're cutting builds up on that ground. It gives more humus and everything and then the roots come through that and so where you're walking, is solid earth and you don't go through [collapsing a Tītī breeding burrow].* (Birder K)

> *Tracks that are too wide and clear can be difficult to catch chicks on during the rama. Chicks will move from nearby fern and low scrub to sit on the track where the birder can harvest them. But if the track is too wide, the chicks will move off them faster to find cover as the birder approaches* (Birder Q).

An exception to the prohibition to cut down live wood is when a tree is about to die, as evidenced by leaves curling and letting sunlight through the canopy (Birder K). Tūpare, one of the main canopy species, has about a 30–50-year lifespan on the Tītī Islands (Birders A, C, L; and see Hawke *et al.* 2003). If the tree will fall and damage the manu, it will be removed rather than allowing it to fall naturally (Birders C, K, R). However, some dead trees have an important purpose. Tītī require a high point from which to jump for take-off. They typically jump from cliff tops or rock promontories, but in some cases they use trees as take-off points. If a take-off tree has died, Birder K will try and retain that take-off tree, and in some cases has propped up individual trees in danger of falling. Birder K has linked the abundance of birds in the area to the presence of these "airports":

> *A muttonbird is a pretty lazy sort of a guy, he won't walk any distance, if he can help it, so they'll use trees and that to climb up [and] to take-off from rather than go to the cliff. If one of those [trees] died you do everything that you can to preserve that take-off, because if those birds haven't got that tree to take off from, they won't bother nesting there. Near the centre of the island there was a big easy manu, which the women used to work. That was a women's manu because it was a nice easy ground to work because there was a nice big tree there for them to take off [from]. It was good birding in that area, there was a lot of birds there, and then the whole clump of trees got that old that the whole lot fell down, and all the leaves went in to those holes and you wouldn't even know that a muttonbird had ever nested there. It was just not a bird or a hole anywhere. Then over the years more trees grew up but they still never nested there. The trees that grew up there was just a thick canopy — tētēaweka just blocks the sun right out, and it wasn't until a big storm broke a big branch off and made a gap, and then they could come up that tree and out through the gap. And then all those birds came back and opened that area up and nested there again. So you never cut or damage one of the airports.* (Birder K)

Birders L and Q mentioned planting of young trees in areas where the old trees died off. Birder P planted young *pūnui* (būnui, *Stilbocarpa lyalli* Gray) in response to damage caused by rats. The pūnui is useful in keeping the breeding burrow entrances free of leaves (Birder P).

Unnecessary disturbance to other birds or vegetation is frowned upon (Birder E, F, L, P, O) and considered to cause the birds to move away, a reflection of a wider concept of reciprocity of care between the birds and the harvesters:

> *... they wouldn't allow you to cut down any trees – you could cut up all the trees that had fallen down, cut those up for wood – but they wouldn't allow you unless it was really urgent to cut down a living thing, because if you did the cycle would just finish. All the plants and that, they wouldn't allow you to kill off anything that you weren't supposed to. That includes the [adult] muttonbirds and all the bird life on the island, you couldn't touch them. You got chastised if you did harm any of the birds or bird life. The only bird life we were allowed to do anything with was wekas and muttonbirds [chicks] because they were things that we had to take to eat and to help with our diet.* (Birder E)

## Keeping the manu tidy

Another method of looking after your manu is to clear wood and debris from the manu (Birders D, F, G, I, L, M, R). In the past, a large amount of wood was removed from the manu and stacked for easy access to be used for firewood. Some birders still cut up dead trees and stack that and other dead wood in piles; however, this practice is not carried out on all islands (Birder O). Birder R recalls his grandparents "cleaning the manu". For some this practice was important in order to make the manu looked cared for (Birder R and L) and it aided access for harvesting. Some also believed this practice promoted access for the birds themselves (Birder G and L):

> *... most people seem to try and keep the manu [tidy], you know stack old stuff that falls over and keep it sort of a bit tidy so that there's some open spaces rather than pushing your way through everything. It's [for] ease of access but I think it's probably better for the birds too 'cause they do struggle in the thick and I think they like the open manu too 'cause that's where they seem to have their burrows.* (Birder L)

A practice that could also be considered "cleaning the manu" is clearing the burrow entrances of leaves. Birders I, M and P cleared out all burrow entrances of leaves and sticks during nanao regardless of whether they thought a chick was in the burrow or not. It is thought this practice might prevent the loss of empty burrows by encouraging birds to use them the following year.

An important "cleanliness" issue is the effective disposal of the entrails and waste products from processing chicks, such as wings and tails. Such waste products are mostly disposed

of in the sea. Some islands with beaches can have this waste washed up on to them. Birders O and P harvest on such an island and they stressed the importance of people not disturbing the gulls that help clean the beach.

## Protection of the breeding burrows

The protection of burrows was the most frequent management practice mentioned. Eighty percent of the interviewees said this was very important. Out of the four birders that did not mention this, three did not take part in the nanao or had not been to the islands since childhood. During nanao, if the chick cannot be reached, a hole is dug over where it is estimated to be to enable the chick to be extracted. This act is called *whakaputa* (Ashwell 1999). It is then important to plug or "*puru*" the hole to ensure the nest will remain dry and the adult bird will nest there again in the next season:

> *They [the old people] have always been pretty conscious of looking after their grounds especially the burrows. If you dig down a puru for to get a bird out you've got to plug that up properly or otherwise you ruin the hole, ruin the nest and the bird is not going to nest there next year. They were very fussy on that one. Don't destroy the holes that's the whole concept you know of not going on the island in the off season and everything else, don't disturb the breeding in any way and don't destroy the holes. And we've always been very conscious of that.* (Birder F)

During nanao, birders are unlikely to harvest chicks on rainy days (Birders C, N, Q, R, S). The main reason for not harvesting in the rain is to avoid getting ill, and thereby missing the rama, which is the more productive harvest period. However, another reason for avoiding harvesting in wet weather was to protect breeding burrows:

> *... you couldn't go out in the rain, because the old people used to say it softens the ground too much and you can't plug the holes properly. So that was the why they wouldn't let them go in the rain.* (Birder C)

## Tapu areas

Designating areas as tapu is used as a way of preventing access to protect areas sensitive to habitat damage:

> *... there's some areas of the island that are tapu, you never ever work them because ground is so soft and the hole[s] are so near to the surface that if you did walk in there those holes would just collapse. ... you didn't go near there, you just left it for the birds to breed in that area, because if you went in there the holes would collapse. ... on a little island like ours whether you like it or not we've got to try and find birds because every hole matters. And then of course the tracks are worked out so that they sort of go round the perimeter of those sort of areas [tapu areas] so that you have a fair chance of picking those birds up at torching time.* (Birder K)

Designating areas as tapu also prevents access to sacred sites associated with their ancestors (*tūpuna*) (Birders C, J, P):

> *... there's a woman and a baby buried there. And mind long before my time, but it's never nanaoed, [and] it's in the middle of my manu. And I've never nanaoed it. I've only known one person to nanao it, that was my brother-in-law, [a] Pākehā, and did he get told off. And it's got all sorts of dry wood on it but nobody goes in to get the dry wood, and even [in] torching time if there's birds on that side, and there always is, I never ever go on to it, no. [I] wait till they come off.* (Birder C)

> *... we don't go where the old houses have been where the old people have lived. You will see rings of burnt rocks and you don't get wood there or anything like that. It's always sort of tapu to us. We just leave them. There are plenty of places you can go and you don't bird there. You don't go and nanao there. I know near where we are there's two big rings of old burnt black rocks where there have been houses and over further there's a patch and it's got all sorts of dry wood but you never ever go in there.* (Birder P)

## Temporal restrictions of harvest — a disturbance reduction strategy

An off-season *rāhui* (ban), centuries old, applies to the islands and serves to prevent disturbance of the ground as well as the breeding adults while they are mating, incubating and, initially, rearing the chicks. As one birder relates:

> *... we were always taught [that] when a bird is nesting, you never go near it. Because that mother may leave the eggs, but once that the bird is hatched, the mother instinct then takes over and it will never leave it. It'll always keep looking after that bird.* (Birder K)

In earlier times, temporal restrictions were also used to minimise the disturbance of the chick's normal emergence behaviour in late April and May, near the fledging stage. The pahure harvest period is viewed as important because it delayed the start of the rama period of night harvesting (Birder D, F, Q). These interviewees believed that if the rama begins too early then the chicks will fledge and leave the island earlier, thereby decreasing harvest opportunities. As one birder explains:

> *... the earlier you go out catching your birds and chasing them, the earlier they are going to jump over the cliff. They are like sheep. If you go out and chase the bloody things every night, they are going to run away as soon as they see you. So if you go out there on the 21st [April] you are starting the ball rolling. So after that when they see your light coming, they are off. I say leave it till later on and go about things quietly.* (Birder Q)

Interviewees reported that if the chicks were frightened back down into their holes, they would stay there until the last possible moment, leaving the island immediately when ready, rather than emerging repeatedly and thereby being available for harvest in the rama period. One interviewee considered that abandonment of the pahure practice was a very important reason for the declines in catches in recent decades.

Birders referred to Rakiura Māori having harvested other petrels on the muttonbird islands in earlier times, particularly *korure* (Mottled Petrel *Pterodroma inexpectata* (J.R. Forster, 1844),) and *parara* ("paras" Broad-billed Prion, *Pachyptila vittata* (Forster, 1777)). Birder L's mother placed a ban on korure harvest for her family's manu in 1949, after detecting a decline in numbers. That rāhui is still in place today.

## Days off: a conservation measure?

Historically, Sundays were always a day taken off from harvesting. In some places, part of Saturday was taken off as well (Birders I, J, E). These were times for rest, social interaction, passing on traditional knowledge, and community

assessment of the harvest, as well as religious observance:

> *... we would work from Monday through till Saturday lunch time and my old tāua [grandmother] she wouldn't let them work after 12 o'clock on Saturday. That was the time that they had for themselves. It was free time and our uncles used to knock off work on Saturday lunchtime, have lunch, have a wash and change their clothing and they'd be away from one end, from Puwai right away to the other end of the island. They'd spend the weekend away and come home again on the Sunday night. Yeah and those at the other end of the island, they'd do the same thing, come back to our end of the island. So yeah they would have a constant change of visitors and they would come and stay the night or just come and visit and go away. In the weekend there was always new faces around, around the house. There would be the old ones, the young ones and there's the young carrying younger ones ...* (Birder E)

As another birder relates:

> *... you couldn't catch a bird on a Sunday and you couldn't go out torching on a Sunday night till after twelve. You weren't allowed to catch birds on a Sunday then. It was religious reasons. They used to have Bible readings, when we were small, every Sunday in the morning.* (Birder C)

Birders F, K and M considered taking days off a conservation measure, because a number of chicks were spared because no harvest took place on that day:

> *And working Sundays, that was a sort of self-imposed conservation, because that was one day, one extra day, you didn't work birds. So many birds [were] saved sort of thing.* (Birder F)

Families no longer collectively take Sundays off. However, they do set aside occasional days for socialising and fishing. On Big Island, Good Friday is a scheduled day off from harvesting and is spent socialising and sharing a meal. These days are usually between the nanao and rama period, when the birders have to wait until the chicks come out of the burrows at night in sufficiently large numbers. At the beginning of nanao, harvest can be delayed for several days if the chicks are deemed too young or small to harvest (Birders C, F, M, N, P, Q, R). The harvest begins when the chicks grow to a suitable size. This decision is made as a group, or by the "supervisor" elected by the group.

In historical times all the chicks were stored using *pōhā* (kelp bags made from the lamina of *Durvillaea antarctica* (Chamisso) Hariot. This required additional days off from harvesting for curing, preserving and storing chicks (Birders C, D, I, M, Q). Once tins (in the 1950s) and then plastic 10-litre pails with lids (from mid-1970s) became the common method of storage, these days off for preparation of the catch were no longer required:

> *... when I was a kid people down there used to have Sundays off and they would have another day off to cut up and they'd have another day off to pack the birds. Today there's no days off. She's all go. So bit of a lost part of the island, you know.* (Birder M)

## Protection of adults

Harvesting of Tītī is solely restricted to the chicks. The adult birds, *kaiaka* (also referred to as *kaiaki*, "mother birds", "parent birds", and "old birds"), are protected because they are the breeders. Killing an adult is considered the ultimate transgression:

> *There is the odd time where your foot will go through the ground and make a hole, [and] if anyone walked on and left that unplugged, that's one of the ultimate sins. That's probably the second worst thing you can do. The worst crime on an island is to kill a kaiaki [adult bird].* (Birder K)

If a child made a mistake in killing an adult they were sometimes made to pluck it (a very difficult task) so that they would be sure to remember to check the age of the bird before killing next time.

## Waste minimisation

An important concept for some of the interviewees was not wasting Tītī or resources on the islands. Over half of the interviewees reported that *kiaka* (a chick that is too thin to take as food) are avoided. Only suitable chicks are killed, and this is done instantaneously so as to avoid unnecessary suffering. Most often the kiaka encountered early in the season are returned to the breeding burrow because the adult may return to feed them and "give them a chance" to survive. However, Birders J and P said that if they found a very weak kiaka they would kill it out of mercy rather than allow the rats to do so. Birders F, G, O, S and Q stated that it is important to harvest only as many chicks as they process. The way the chick is handled through the catching, processing and storage stages is important in order not to ruin the bird (Birder K). Chicks that become ripped, bruised or deemed unsuitable for sale, barter or gifting become "home" or "tucker" birds and are retained by the birders for their own food (Birder R, G, F, I).

## Sanctions, compliance and enforcement

"Clubbing" has been banned on some islands and manu (Birder F and M). "Clubbing" involves the use of a small club to kill the chicks during rama. The main objection to this method is concerned with the possibility of wastage, as this method eliminates the need to handle chicks before killing them, introducing the possibility that small chicks and adults could be mistakenly killed. Bruising of the meat can occur, which makes it less suitable for sale. "Fencing", where Tītī are trapped during rama when they make their way to cliff take-off points, is another method that has been informally prohibited, because of the risk of wastage and harming adult birds. Both clubbing and fencing were formally banned by new bylaws passed for the Rakiura Tītī Islands in 2008, but community sanctions based on tikanga had virtually eliminated the practice for several decades on all the islands.

Access to the Tītī Islands is strongly protected. Beneficial owners have removed potential owners from the island, or prevented them from taking part in the harvest, if they damaged the island or misbehaved (Birder E and P; see also Wilson 1979). As one birder recounts:

> *... one year some [of] my cousins came down, [and] they didn't know anything about muttonbirding and they were just starting at the mouth of the hole and digging their way back until they found a bird. 'Cause they'd never been told and I remember they were told if they couldn't do it properly, [then] stay out of the bush. They weren't allowed to go out and that then. If you don't respect the island and things down there, well they always get told off and if you don't do it, the owners will get together and tell you to keep off the island.* (Birder P)

The tikanga or rules that protect the islands are debated fiercely at the annual Permit Day hui and any transgressions are noted and publicly aired. The rules are viewed as the one thing that the birders have direct control over and must protect at all cost to protect the birds and habitat:

> *I think the habitat and how we look after it is really important for the returning birds when they come back, because no matter what happens out there we're the only ones who have got control of the habitat 'cause there's no control in the ocean. There's so many things happening out there that we have got absolutely [no] control [over] unless we can convince Governments to stop pouring out pollution and all the things that have adverse effects in the ocean. So you know, we should do everything we can on the land to make sure that the ones that do come back have the very best habitat there is for them.* (Birder L)

### Reciprocity, responsibility and learning for sustainability

Birders believe that if they look after the manu, the birds will keep coming back and will continue the harvests for the generations to come:

> *I think how you look after an island dictates as to what will be on that island, because those birds keep coming back to that island. I know that the mothers do. Once they've nested, they will keep coming and nesting there again ...* (Birder K)

In some of the interviews, the kaumātua and elders mentioned a responsibility they had to look after the islands for the next generations and that it was important to pass on the knowledge and the resource to the next generations. This responsibility was given to them by their ancestors. This knowledge had been passed down to them from their parents or (in many cases) grandparents. Approximately a quarter of the birders stated that it is important for the next generations to be taught properly to look after the islands for the sustainability of the Tītī and the cultural heritage of Rakiura Māori. Looking after the land is also looking after your people:

> *I think kaitiakitanga is really looking after what is here now and making it improve and grow. I see kaitiaki[tanga] as being what we have done on [our island]. I see it all being part of the lifestyle of the people on the island. But I [also] see it too as how we look after our families. Because that's kaitiaki[tanga] too. We are kaitiaki of those children and young people. Then it transfers on to the land. I see what we have done down there, how the island's starting to blossom, I see that being kaitiaki[tanga].* (Birder L)

Another birder relates:

> *... most of the people I know down there [on the Tītī Islands] look after the muttonbirds. There's the odd one in every situation that don't — that are grabbers and don't care about the holes or the way the ground's left or anything. I have seen it myself. But the majority of them, thank God, are good conservationists. As long as that happens and you teach the young ones properly so that they come along to be good conservationists and look after that what you got, then you got no problem. I don't and the [Forest and] Bird Society should not worry about it. Because if you have got to be told by somebody else how you should run something that you have had for generations, you're at the end of your tether. So you might as well stop.* (Birder M)

Many birders emphasised the importance for the next generations to be taught properly to look after the islands for the sustainability of the Tītī and the cultural heritage of Rakiura Māori (Birders J, L, K, M, N, S).

## DISCUSSION

### Mātauranga and tikanga for sustainable resource management

The Tītī harvest provides an important case study of TEK and common property theory in action within New Zealand. However, calls for reinstatement of customary harvest rights around the world (Posey 1996, IUCN 1997), and ensuing debates about the environmental safety of indigenous people's harvest management, broaden the conservation interest of this study. Our central inquiry was whether resource management rules exist to secure the sustainability of a culturally-defining customary harvest for the mokopuna.

Rules dictate the behaviour of Tītī harvesters in relation to the resource. For example, rules ensure adult birds are not disturbed, only chicks of a suitable size are harvested and that wastage is minimised. However, the majority of rules focus on the protection of the habitat of the island, such as burrow integrity, care of vegetation and maintenance of manu areas. The potential threat to adult birds and possible wastage has prompted the development of rules banning certain techniques for harvesting Tītī. These rules fit under the "method taboo" category of Colding & Folke (2001). Similar rules are applied by Hauraki Māori when harvesting Grey-faced Petrels, *Pterodroma macroptera gouldi* (Hutton, 1869), in northern New Zealand (Lyver *et al.* 2008). While differing in specifics to match local ecology and behaviour, these rules fit into the categories established for other cultures' sustainable use of a common resource (Folke *et al.* 1998, Colding & Folke 2001, Berkes *et al. 2003*).

Some Tītī harvest rules obviously serve several different functions at once and sometimes they were not justified or promoted for solely "conservation" or "sustainability" reasons. Indeed, we were struck by how infrequently the words conservation, sustainability and kaitiakitanga were mentioned by the interviewees. The mātauranga and tikanga described to us was about the practice of birding and usually only secondary reference was made to their sustainability or conservation consequences. Similar observations of other indigenous communities have led some Western observers to discount lore as being a conservation mechanism, simply because it is not always enunciated as being directed or wholly-designed for a conservation purpose (Smith & Wishnie 2000). This *etic* (outsider) perspective is tautological and based on cross-cultural presumptions about the design of social institutions. The more important debate is whether the lore protects the natural resource, harvest sustainability and social-ecological resilience, to deliver that harvest sustainability in the long term.

Adherence to the tikanga is fiercely reinforced within the Rakiura Māori community, both for managing the birders and controlling outside groups like researchers. For example, prohibition of our field ecology researchers visiting the manu early in the season was firmly enforced (Moller 2001a, Newman & Moller 2005). The researchers required estimates of the number of eggs laid and the proportion

that hatch and survive until the early chick phase (when the birders are not allowed to "walk on the manu") for the construction of computer models to estimate harvest sustainability. The researchers were therefore forced to measure the early season population parameters on other non-harvested study areas. Similarly, for a decade the researchers were prohibited from digging up burrows to check the accuracy of the burrowscope method used for estimating chick density and breeding success (Lyver *et al.* 1998, Newman *et al.* 2008 a, b, Scott *et al.* 2008). The kaitiaki reluctantly permitted the researchers to dig a very small number of plots with breeding burrows only when all other methods to check the accuracy had been exhausted (McKechnie *et al.* 2007). Even then digging was focused as much as possible on sites where new buildings were soon to be erected (the burrows on those sites were soon to be disrupted anyway), and the very laborious traditional puru technique had to be used to reseal the burrows after plots had been surveyed. These cultural restrictions on research are akin to the ethics guidelines operating in wider society, but crucially different in their adherence to local tikanga. A third example of cross-cultural differences in research ethics related to researchers' requests to attach radio-transmitters to adults. Even though the transmitters were within internationally recognised research standards for size and attachment method, the kaitiaki were extremely sceptical that they would not disturb the birds and insisted on a trial using dummy transmitters in a colony away from the harvested manu before they would authorise more widespread telemetry. To the researchers' surprise, the trial did find evidence of disruption to colony attendance from having the transmitters attached (Söhle *et al.* 2000), and wider use of telemetry was abandoned after the initial six deployments (Söhle *et al.* 2007). Matching research ethics to local tikanga retained community confidence in the research and minimised disturbance to the habitat and adult birds, but scientific inferences were weakened and the culturally appropriate alternative research paths were more expensive and less powerful. More importantly in the context of this discussion, the tikanga and mātauranga of the birders was demonstrated to be alive and well, vigorously asserted and, in the case of the telemetry example, both accurate and valuable.

Tikanga stipulating an imperative to only take what can be processed is very important for sustainability because the time available for harvesting and processing the chicks places a direct limit on the number of Tītī harvested. With the current technology, the amount of time it takes to process the chicks limits the number of chicks that can be harvested without wastage (Lyver & Moller 1999a, b, Lyver 2000b, Kitson 2002). There are no rules that dictate the number of Tītī birders can harvest from a particular area, or on a given day, per season, or per person participating. This finding is consistent with other studies of wildlife harvests by Māori in New Zealand (Moller & Lyver in press) and of indigenous resource users in general (Wilson *et al.* 1994, Colding & Folke 2001). It leads to many observers characterising indigenous harvest management as "passive", "compared to active" management of the size of the take itself that is normally promulgated by Western wildlife management regimes. However, our research recorded many and varied active rules by which sustainability is promoted indirectly.

One of the most important harvest rules is the protection of breeding adult birds, which has also been highlighted as the most vulnerable life stage to harvest for long-lived seabirds. Seabirds are long-lived and have low productivity, so the demographic impacts of harvesting chicks are much less than if adults are taken (Lyver & Moller 1999b, Hunter *et al.* 2000a, Moller 2006, Moller & Lyver in press). Tikanga prescribing that the very small "kiaka" chicks should be left will have negligible impact on sustainability towards the end of the season because these starved chicks do not survive anyway (Sagar & Horning 1997, Hunter & Caswell 2005). However, returning the smaller chicks to their burrows early in the nanao period does potentially reduce harvest impacts because feeding by the adults in the last stages of the fledging period can rapidly increase chick size, thereby lifting more of the chicks over the threshold where they may survive to recruitment stage.

The rāhui preventing access to the islands until late in the chick phase is probably particularly important for reducing breeding failure and emigration by pre-breeding birds. Scientists have also found that Tītī are susceptible to disturbance in the egg phase (Warham & Wilson 1982). Rapid decline of the Fisher Island population of Short-tailed Shearwater was apparently triggered by repeated handling and disturbance at the colony in early and mid-stages of the breeding season (Serventy & Curry 1984). This sensitivity to disturbance is believed to be a common phenomenon in procellariids (Warham 1996).

Tapu is a central concept in the Māori culture (Metge 1976, Marsden 1981, Irwin 1984, Patterson 1994). Everything designated as tapu must either be avoided or handled with care according to prescribed rules (Metge 1976). Tapu is normally understood as prohibition (Irwin 1984), but in resource management its function can be for protection (Irwin 1984, Puia 1990). Declaration of tapu areas can be important for protecting resources and habitats in some areas. For example, the ground on one of the small Tītī Islands is extremely fragile and susceptible to burrow collapse because the native Tūpare has been displaced by a shallow-rooting invasive tree species. However, on most islands the tapu areas are small. Also, all tapu areas will only provide a partial refuge from harvest for chicks, as some chicks will move out of these areas just before fledging and may therefore be captured.

Overall, only 15% of the Sooty Shearwater breeding ground within the Rakiura Tītī Islands is not harvested (Newman *et al.* 2008a). This area mainly comprises small islets that are either too steep or too small to support a whānau for birding. Within the birded ground, additional protection by tapu is very limited and can thus be excluded from consideration in quantitative assessments of the sustainability of the harvests, at least as a direct effect. However, observance of tapu will have a much greater indirect effect on Tītī harvest sustainability by reinforcing reciprocity and observance of tikanga, especially the very important *rāhui* provisions (protection of adults, banned access to the islands early in the breeding season). Scientific reductionism that separates rāhui from a wider context of tapu, reciprocity, identity and place will obviously fail to recognise a whole package of interrelated tikanga and world view that promotes sustainability in indirect ways.

## Access, compliance, lore and law

Rakiura Māori have both informal and formal institutions to govern their behaviour. Social mechanisms (informal institutions) can be far more effective and are less costly as

compliance and enforcement mechanisms because of their self-regulatory nature (Colding & Folke 2001). For common property resource management to work, it is important for the resource users to believe that abiding by the rules will bring benefits to them as individuals. Effective self-regulation is expected to be especially important in managing use of resources in remote areas like the Tītī Islands, so this is a common motivation for instituting co-management in such places (Moller 1996, Taiepa *et al.* 1997, Moller *et al.* 2000). In particular, social sanctions function as guides for human conduct towards the natural environment (Colding & Folke 2001).

While the harvest had been governed by traditional rules, in 1910 Rakiura Māori petitioned the New Zealand Government to recognise most of these rules as legal regulations (Land Act Regulations 1912, Department of Lands and Survey 1978). This legal recognition was seen as a way to negate any changes to island land use and open access to others, for example leaseholds and farming (Wilson 1979, Birder L). There was also concern that the Tītī resource would be exploited by northern Māori. Accordingly, access to the Tītī Islands became strongly protected by law in a way that reflected lore.

Reinforcement of lore and law has been an ongoing and collaborative effort of the Rakiura Māori community and government. At Permit Day hui, permission for access goes through close community scrutiny and sometimes-heated debate. In the past, trespass has been penalised by confiscation of harvest and public notification of the offence. Pākehā have also been prosecuted and penalised for trespass on the islands (*Southland Times* 3 July 1913). Historically, blood has been spilt in fights between island owners and trespassers on the islands (Wilson 1979). The testimony of our interviewees and government files (formerly of the Lands & Survey Department; then Department of Conservation) record repeated and ongoing debates about who does and does not hold harvest rights and instances of community interventions to reinforce access rules. These access rules are upheld just as vigorously today. For example, in the late 1960s, one interviewee was stopped at the wharf just before embarking to the island because he had not gone through the Māori Court and gained a succession order to allow him to go to the island. Bell (1962) recorded discussion about regulations not being followed in all their details. Correspondence on Lands & Survey files in the early 1970s report concerns that the supervisors were not asserting their traditional authority or that birders were not heeding it, so the Commissioner of Crown Lands for awhile visited during the birding season, accompanied by the supervisors, to provide a more visible presence on islands. In 2003, at the request of beneficial owners, the Department of Conservation removed non-beneficiaries from two manu in 2003 (S. Owen, pers. comm.). In 2004, a photographer was forcibly removed from the islands because he was not a Rakiura Māori, even though he was being hosted by a birder with rights. The eyes and ears of the birders effectively police the access to the islands and use of the law to reinforce lore is obviously frequent and effective. This reinforces a strong sense of community agency and responsibility.

There has been repeated reference in Permit Day hui over the past 14 years of research to the way the muttonbirding laws were requested, designed and reinforced by the tūpuna. The birders see the legal regulations as their own regulations and consider that they must be protected and reinforced continually. In contrast, some Rakiura Māori individuals talking on the marae have occasionally reported their transgression of Resource Management Act and Conservation Act laws applied to management of other resources on the mainland, or even expressed some pride in transgressing where those laws are seen as nonsensical or not reflecting lore.

While the muttonbirding regulations broadly reflect traditional lore, some of the provisions clearly serve a more bureaucratic and surveillance role for government. Legal regulations instituted powers on "supervisors" which traditionally were exercised by kaumātua. Supervisors are appointed for each island at Permit Day hui prior to the season. They have legislated roles in settling disputes during the birding season, reinforce the regulations and provide an annual summary of the total number of chicks harvested from their jurisdiction. Several birders have indicated to the researchers that the latter are widely disregarded and inaccurate. Birders closely guard their harvest tallies from each other, so a legislated requirement to declare the level of catch is largely ignored. The above examples indicate that simply regulating new requirements, like supervision and harvest reporting, is unlikely to succeed if not embraced by the majority of harvest practitioners themselves. If accepted by the majority, law can then be exercised to enforce compliance of miscreant individuals.

There is potential for law to reinforce traditional social structures, such as hierarchies based on age, but substitution of traditional authority by law by itself is unlikely to empower self-regulation for sustainability. Elders play a key role as the keepers of TEK, transmitting knowledge and providing wisdom to interpret novel observations (Berkes 2008). That respect of elders for their knowledge and authority is still strong in the Rakiura Māori community is reflected in the way the community directed the researchers to seek the testimony of the kaumātua for this research. However, some interviewees expressed concern that younger people were no longer respecting their elders and the lore (Moller & Kitson, unpublished data) and one kaumātua goes further to assert that some of the older ones are actually frightened of some of the younger community members (Bragg & Newman 2004).

Incorporation of Christian beliefs into Māori tikanga is widespread and reflected in recitation of Christian prayers in many community activities. In some instances, such beliefs have even been transferred into law. For example, regulations applying until 1962 stated that: "No work in connection with the taking or preserving of the muttonbirds shall be done before 6 pm on any Sunday" (Land Act Regulations 1949, amendment No.3 (1962)). Amendment of law is time-consuming and expensive, so it is perhaps inevitable that aspects of the law will not closely match current lore. The way law affects customary harvest management is a potentially important and unstudied aspect of adaptive co-management of natural resources by indigenous peoples throughout the world. In which circumstances will it help retain traditional management methods and protect sustainability, and in which might it hinder?

While instigation of legal regulations to reflect lore can be seen as promoting sustainability, it could never be a complete substitute for the tikanga and detailed knowledge that is continually being exercised to protect the birds and the breeding habitats. The past or current regulations do not cover protection of vegetation, yet the majority of interviewees stressed the importance of this. This indicates the

presence of social sanctions that are still guiding behaviour of Tītī harvesters at a much more subtle and detailed level in ways that will be very important for sustainability.

## Common property resource management theory

The Tītī and the land on which they breed is communally owned and managed. Common property theory operates where the exclusion of potential users is difficult and the activity of each user subtracts from the welfare of all the others (Berkes 1989, Feeny *et al.* 1990). It might seem that the remoteness of the Tītī Islands should afford a level of protection by excluding other potential users. In fact, journeys to the islands in canoes were undertaken for centuries (Davis 1999) and in contemporary times commercial fishers regularly land to take "lunchtime birds" during the season (J. Kitson & H. Moller, unpubl. data). The promulgation of the muttonbirding regulations was partly motivated to protect the resource from North Island Māori and non-Māori, but strident debate at annual Permit Day hui throughout this study period has repeatedly focused on access issues, associated "birding rights", and reinforcement of boundaries between manu amongst Rakiura Māori's own community. Government correspondence files demonstrate that these have been ongoing concerns since the early 1920s. Notwithstanding these debates, it is widely believed that by adhering to the rules, which are reinforced by legislation and cultural and social sanctions, the birds will return, providing an incentive for individuals to comply. This study therefore confirms that Rakiura Māori Tītī harvesting practice is an example of common property resource management. The collective international empirical evidence of the efficacy of common property law for resource protection (Berkes 1989, Feeny *et al.* 1990, National Research Council 2002) therefore gives us confidence to assert the power of the tikanga to create the necessary platform for sustainable management of Tītī.

## Adaptive co-management and learning for sustainability

Adaptive co-management emphasises the importance of learning and creation of new social institutions to meet new sustainability challenges (Berkes & Turner 2006, Turner & Berkes 2006). The experience and wisdom of TEK holders provide the context for this flexibility in rules (Hviding 1998, Berkes & Folke 2002). An important aspect for enduring cultural and ecological sustainability is the mechanisms by which TEK is received, accumulated and then transmitted to the next generation (Ohmagari & Berkes 1997, Folke *et al.* 1998).

Our study has shown that rules, sanctions and harvest organisation can be different on different islands. The harvesters reorganise and adapt to new situations and circumstances while still keeping within the generally-defined bounds of acceptable social behaviour. Turning lore into law is a dramatic cross-cultural adaptive co-management adjustment. Application of ecological science alongside tikanga and mātauranga for sustainability is another powerful example of adaptive co-management by the kaitiaki of the Tītī. Although the majority of the community supported and therefore chose to go ahead with the *Kia Mau Te Tītī Mo Ake Tōnu Atu* research program, some individuals opposed it throughout its 14-year duration. One interviewee in another study using confidential questionnaire approaches summed up his concern by saying "beware of strangers bearing gifts" (Moller 2003). The predominating concern was that the research brought external scrutiny and therefore the possibility of external interference in harvest management.

The Rakiura Māori community have also adapted to embrace many other contemporary tools and institutions for wildlife management and island ecosystem restoration. Tītī harvesters on the eastern Tītī Islands (fig. 1), which are regularly re-infested by rats from the main Rakiura island, combine efforts and resources in an attempt to control rat numbers (Birders P and H) by laying rat poison. Birders combined forces with the Department of Conservation to eradicate Kiore from Putauhinu (Davis 2004). Following the drowning of a banded Tītī in a 1998 oil spill off San Francisco, the birding community successfully petitioned an oil spill reparation fund in the USA to pay for eradication of Ship Rats and Kiore and more recently Weka from four Tītī Islands (Moller *et al.* 2003, Coote & Blackwell 2006) Eradication of introduced predators has enabled transfers of threatened bird species, such as Codfish Island Fern-birds, *Bowdleria punctata wilsoni* Stead, 1936, and led to a resurgence in plant, insect and other bird life, helping to restore the ecosystem. The community's adoption of these international and contemporary wildlife management and research tools was combined with their local knowledge, voluntary effort and community authority to reduce rat infestation from 48% to 17% of the Tītī breeding area.

## Reverence for life, reciprocity and learning for sustainability

Our research adds to a growing body of international case studies (reviewed by Berkes 2008) to show that indigenous societies' resource and habitat protection rules function similarly to westernised conservation management. For example, habitat protection is a large part of westernised resource management and Rākiura harvesting rules and the tikanga minimise disturbance, protect breeding stock and sometimes promote escape from harvest in just the same way as scientific management. Some of the parallels found in the results of international studies reflect common ground between the approaches of different cultures to environmental management. This is likely to represent convergent cultural evolution of different knowledge systems to meet common challenges of ecology, social-ecological resilience and sustainable resource extraction. This convergence arises despite notable differences in the specific ecology and behaviour of the hunted species and social institutions operating in divergent cultures and communities.

An overarching Māori world view emphasises interconnection between people and their environment and between current generations and their ancestors as part of a wider reciprocity (Roberts *et al.* 1995, Patterson 1994, Kawharu 2002). Threaded through the testimony of the kaumātua we interviewed was a fundamental concept of reverence for life and reciprocity. There have been many instances over the 14 years of this study where non-Māori have communicated their distaste of Māori harvesting wild and/or native animals, and especially their chicks. These observers consider muttonbirding as showing disrespect for nature and life, tantamount to sacrilege. However,

the Tītī harvesters asserted many rules for promoting life on the islands and profound reverence for the Tītī, their beauty and strength. Simple direct examples include the staunch rules such as protection of living trees and all adult birds. Some birders practise mercy killings of starving chicks in much the same way that animal welfare ethics are applied in wild food harvesting and agriculture (Matthews 2008).

More fundamentally, many birders confidently assert a general belief that harvested populations are more healthy and ecologically resilient. Similar constructs are found in the mātauranga concerning other customary harvests of wildlife by Māori (Moller & Lyver in press). Efforts to clear the manu and keep it tidy were expected to promote overall abundance of the birds. In the early phases of "bringing in" a manu when it is first worked, some birders deliberately shorten breeding burrows in the belief that this would promote the number of birds breeding in the area. This was mentioned to the research team by other muttonbirders during the 14 years of study (H. Moller, unpubl. data), but none of our interviewees mentioned it, presumably because their manu had been long established on traditional ground. Similarly, Richdale (1944) reported that the 1942 birding season was very poor and that the birders believed this was because the burrows had not been cleaned out because birders had not visited the islands the previous season (World War II disrupted normal birding). Populations with greater resilience might result from density dependence, the main ecological mechanism by which harvest off-take could be compensated (Moller 2006), but this would require reduction in local density. There is tentative evidence of density-dependent regulation in the Tītī population, in part triggered by improved breeding success in less densely burrowed ground (Newman *et al.* 2008a, in press).

Reciprocity between the harvesters and the birds can be approached by the ecological linkages, such as the postulated density dependence mechanisms. However, a deeper and more fundamental recognition of reciprocity underlies the expectation of some birders that the Tītī will move away if they or their habitats are disrespected. Some Tītī harvesters drew our attention to a deep relationship between the birds and the people. These birders consider the birds have feelings and their own agency in that relationship. For example, some birders expressed their belief that the birds "called the researchers to us [the *Rakiura* community]" to help arrest declining bird numbers (e.g., see the discussion by Tane Davis in the *Whakamutunga o Te Hui* (farewell) session in Howard & Moller 2001). Other birders, who have been sceptical of the value of the *Kia Mau Te Tītī Mo Ake Tōnu Atu* research program, have expressed their opinion that an upsurge in numbers and condition of the birds in 2007 and 2008 seasons was a direct result of the field research ending in 2005 (R. Clucas, unpubl. data). One of those birders stated that the birds have felt a "weight lifted off their shoulders" by the research ending. These are examples of some kaitiaki seeking support and guidance from the birds themselves. It is a fundamentally different construct from Western wildlife management approaches where broadly scientific decision making operates. Both approaches are motivated by care, but science is nevertheless applied to manage birds that are considered passive recipients, whereas some birders speak of a two-way interaction between people and the birds that operates at a metaphysical level. This is an example from the Tītī case study of the "belief" component of TEK's knowledge-practice-belief nature (Berkes 2008). Metaphysical beliefs cannot be confronted on science terms but nevertheless must be acknowledged in science and mātauranga partnerships being directed by an indigenous community level (Moller 2001b).

Transgenerational reciprocity and responsibility was also emphasised repeatedly by the kaumātua of the Tītī community. Current kaitiaki expressed their responsibility to uphold the tikanga and continue the mātauranga because it was required of them by their tūpuna as much as by their mokopuna. Transgenerational equity is at the core of the conservation ethic of all cultures and ethnicities. However, the inclusion (some kaitiaki would say participation) of the tūpuna in the dialogue and decision making of the Tītī community has been frequent and emphasised. This inclusion makes discourse and decision making in the community fundamentally different from that we have experienced in scores of forums concerning harvest and conservation management by Pākehā over the past 14 years since the *Kia Mau Te Tītī Mo Ake Tōnu Atu* program began.

## Environmentality and kaitiakitanga

It is important that Rakiura Māori retain the control of Tītī harvesting, both for the Tītī *per se* and for their cultural wellbeing. Societies do not establish conservation rules and ethics for the benefit of outsiders (Berkes 2008). Incursions of outsiders and the inability of a group to defend an important resource have been shown to cause the lifting of rules and the decline of conservation ethics (Feit 1986, Berkes 1986, Berkes *et al.* 1989). The very name given to the research program, *Kia Mau Te Tītī Mo Ake Tōnu Atu* ("Keep the Tītī forever"), was repeatedly referred to as a double imperative, that is, to ensure that the birds remain plentiful enough for the mokopuna but also for the kaitiaki to retain management control of the islands and the harvest. The provisions of the cultural safety contract guiding research co-management were mainly designed with the second objective in mind. The experience of partners in environmental co-management (Moller 1996, Taiepa *et al.* 1997, Moller 2001a) and the emerging theory of environmentality (Agrawal 2005) focus on appropriate governance institutions to trigger individual transformations amongst resource users' subjectivities about the environment. Promotion of the agency of locals, sharing authority and devolution of power to create meaningful roles in resource monitoring and harvest regulation are seen as key to building responsibility and voluntary compliance. The effectiveness of subsidiarity and bottom-up approaches to conservation management relies on individual commitment to local community goals, especially when resources are communally-owned and harvest activity may be relatively solitary and remote from the scrutiny of others. The reality is that effective external management of the Tītī harvest by traditional wildlife management lore would be practically impossible. The theory of environmentality predicts that imposing rules from outside the culture and community would also be fundamentally undesirable. In contrast, Tītī harvesters have a strong sense of ownership of the tikanga and rules, which they believe must be upheld at all costs, in order to protect the birds and habitat.

## *Ahi kā roa*: keeping the fires burning for the mokopuna

Integration of seasonal movements and hunter gathering, maintenance of knowledge and occupancy over successive generations is referred to *as ahi kā roa* (keeping the fires burning) by Māori (Roberts *et al.* 1995). At the final community hui where the researchers presented Tītī sustainability predictions from science to the Rakiura Māori community (Murihiku Marae, 5 July 2008), a kaumātua summed up his assessment of the science as "only numbers" and followed by emphasising the need to reassert the tikanga and find ways for the mātauranga to be transferred to the younger generation (Moller 2008). This is linked to the issues of pride and responsibility, many Māori would say *mana* (prestige), as environmental stewards and being able to sustain and provide resources for their own people and *manuhiri* (visitors) (Waitangi Tribunal 1991). Birder M summed it up nicely as: "if you have got to be told by somebody else how you should run something that you have had for generations, you're at the end of your tether. So you might as well stop". Like many of the simple and direct comments of the Tītī harvesters, this is a powerful statement of the inter-relationship between identity, agency, responsibility and sustainability.

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# A POPULATION AND HARVEST INTENSITY ESTIMATE FOR SOOTY SHEARWATER, *PUFFINUS GRISEUS*, ON TAUKIHEPA (BIG SOUTH CAPE), NEW ZEALAND

by Jamie Newman, Darren Scott, Henrik Moller and David Fletcher

(with one text-figure and two tables)

**Newman, J., Scott, D., Moller, H. & Fletcher, D.** 2008(31:x): A population and harvest intensity estimate for Sooty Shearwater, *Puffinus griseus*, on Taukihepa (Big South Cape), New Zealand. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 177–184. ISSN 0080-4703. Centre for the Study of Agriculture, Food and Environment, University of Otago, PO Box 56, Dunedin 9054, New Zealand (JN, HM*), 30 Hill Road, Purakanui, Dunedin (DS), Department of Mathematics and Statistics, University of Otago, PO Box 56, Dunedin 9054, New Zealand (DF). *Author for correspondence. Email: henrik.moller@otago.ac.nz

We estimated the total number of burrow entrances, chicks and total population size of the Sooty Shearwater, *Puffinus griseus*, colony on Taukihepa (Big South Cape), the largest of the 36 Titi Islands where Titi (Sooty Shearwater chicks) are harvested. Between 1999 and 2005 we surveyed six manu (family birded areas) as well as a colony in the interior forested region on the island, measuring entrance density and burrow occupancy and habitat variables, as well as recording birders' harvest intensity. We found little support for any relationship between habitat variables and entrance density on the surveyed manu. Therefore we used a detailed aerial image to map areas and extrapolate these estimates to island totals and a simple population model, incorporating known demographic parameters to calculate the total population size. We estimated that the island contains 1.67 million (95% CI: 1.47–1.88) burrow entrances and 1.12 million (0.959–1.28) breeding pairs, equating to a total of 5.31 million (95% CI: 3.98–6.77) juveniles, pre-breeders and adults associated with the island. Assuming the harvest intensity on the surveyed manu 0.18% (0.16–0.21) was similar elsewhere on the unsurveyed but harvested region of the island, we estimated that of the 807 000 (712 000–901 000) fledging chicks on the island, 137 000 (105 000–173 000) are harvested each year. These findings highlight the importance of the island both in terms of total Sooty Shearwater population in New Zealand and to the Rakiura Maori who harvest the Titi.

**Key Words: *Puffinus griseus*, Sooty Shearwater, Titi, Taukihepa, Big South Cape, New Zealand.**

## INTRODUCTION

Taukihepa (Big South Cape Island; 790 ha, 47°14'S, 167°25'E; fig. 1) is the largest of the 36 Tītī Islands that surround Rakiura (Stewart Island) in southern New Zealand. In terms of total vegetated area, it represents 54% of all the Tītī Islands (based on two-dimensional planar area calculations) and is known to contain one of New Zealand's largest populations of Sooty Shearwater *Puffinus griseus* (J.F. Gmelin, 1789) (also known as Muttonbird, the chicks commonly referred to as Tītī). Tītī on this group of islands, including Taukihepa, are exclusively harvested by Rakiura Māori, New Zealand's southernmost Māori community, as one of the few large-scale customary harvests remaining in New Zealand today (Wilson 1979, Taiepa *et al.*1997). The sustainability of this harvest is the subject of a long-term research partnership between Rakiura Māori and the Department of Zoology at the University of Otago (Moller *et al.* 1999). A key component of the *Kia Mau Te Tītī Mo Ake Tōnu Atu* ("Keep the Tītī forever") program is the assessment and ongoing monitoring of Sooty Shearwater populations, both in areas with and without harvesting in order to detect spatial and temporal trends in abundance.

In addition to being of great significance to the Tītī harvesting community, Taukihepa has great conservation potential as an offshore island refuge. Taukihepa became internationally infamous in conservation circles in the 1960s for an ecologically devastating invasion of Black Rat, *Rattus rattus* (Linnaeus, 1758), that resulted in at least three species extinctions and severely depleted the populations of many of those species that remained. This devastation is one of the most recent and salient examples of the rapid damage an introduced mammal can inflict on a naïve flora and fauna – although the total effect was poorly monitored and documented. Despite the publicity surrounding the rat invasion and the apparent importance of the island, relatively little has been published about the island's flora (Fineran 1973) and fauna, including any information on the abundance of Sooty Shearwater. Burrowing seabirds are recognised as "keystone" species on many offshore islands (Maesako 1999, Sanchez-Pinero & Polis 2000). They are known to massively elevate the input of nutrients from the marine environment (Hawke & Newman 2004), as well as increasing levels of biopedturbation (McKechnie 2006), influencing a wide range of ecosystem processes (Fukami *et al.* 2006). As large sections of Taukihepa have now been extensively surveyed as part of our research program, we are able to report for the first time a Sooty Shearwater population estimate for the entire island based on robust survey techniques.

We also researched the relationship between burrow entrance density and vegetation to see whether information on habitat could improve the overall estimation of population density both on Taukihepa and elsewhere.

This paper presents data on the size of the colony and the density of the breeding burrows and occupants – leading to an overall estimate of the total number of burrow entrances and Sooty Shearwater chicks on Taukihepa. It describes the relationship between entrance and chick density and habitat variables, and provides an estimate of the number of chicks harvested each year from the island. It also provides an overall estimate of the total population of Sooty Shearwater on Taukihepa, by extrapolating from our total chick estimate using available information on relevant population demographic parameters. In addition, it provides a population baseline just prior to the recent eradication of *R. rattus* from the island (Coote & Blackwell 2006).

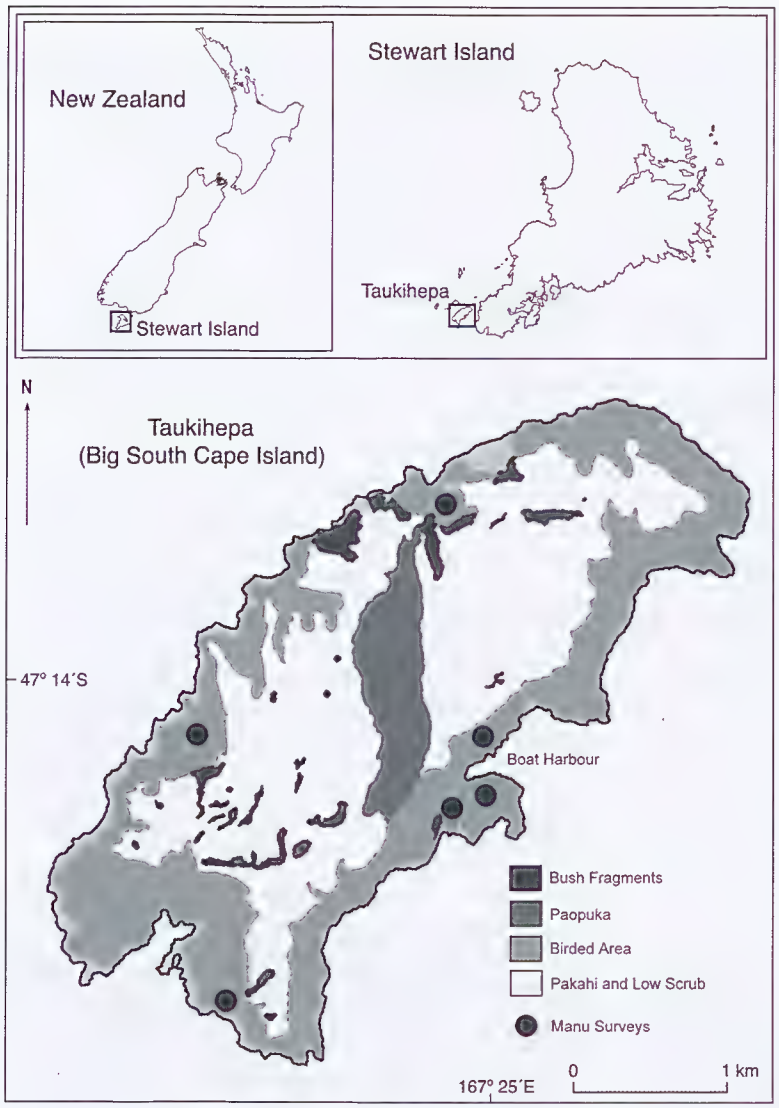


*FIG. 1 — Map of Taukihepa showing the vegetation types used in this study and the approximate location of the six manu surveyed.*

## METHODS

### Sampling design

As part of the *Kia Mau Te Tītī Mo Ake Tōnu Atu* Research Project six *manu* (spatially distinct family birding areas) were comprehensively surveyed to estimate burrow entrance and chick density between 1999 and 2005 (Newman *et al.* 2008). All of these surveys were carried out between 15 March and 4 April, and always took place prior to any harvesting occurring on the manu.

Each individual manu survey was carried out using a stratified-random sampling design following the same standardised protocol. Typically the manu was split into 8–12 sections based within which 12–24 circular plots and one or two permanent transects were randomly located. The location of section boundaries and transect locations were recorded using handheld 12-channel GPS units (Garmin GPS12). Two-dimensional (surface) areas were calculated from these points using GIS software (MapInfo Professional v6.0) and aerial survey photographs taken in 1976 (Series: K/18 SN 5003) and 1988 (Department of Conservation 1:4, f =50 mm, flight elevation 10 000 feet). Each manu survey typically took four people about 6–8 days to complete.

### Measuring burrow entrance density

For each circular plot a limited set of habitat variables (table 1) was recorded and the number of burrow entrances within a 3 m radius plot recorded. In total 1041 circular plots were measured on the six manu surveyed.

### Measuring burrow occupancy

Transects were extended from their random start point on a random bearing until 20 burrow entrances were detected within a 1 m-wide strip (2 m wide in areas of low burrow entrance density) either side of the centre line. Each of the 20 burrow entrances were then prospected using an infra-red burrowscope (Lyver *et al.* 1998) to determine occupancy. In addition to recording burrow occupancy, a number of detailed habitat and environmental variables was scored along the length of the transects. A subset of these was used for this analysis (table 1). In total 77 transects were established in the six manu surveyed on Taukihepa.

**TABLE 1**
**Summary of variables measured within circular plots and transects**

| Variable | Description | Mean (± SE), Median (range) |
|---|---|---|
| ***Circular Plots (n = 1041)*** | | |
| % Ground Cover | Proportion of the plot covered in low level foliage (up to waist height) | 36.1 (± 1.1), 20 (± 100) |
| % Canopy Cover | Proportion of plot covered with some type of canopy above waist height | 78.4 (± 0.8), 90 (± 100) |
| *Composition of canopy* | | |
| % Tūpare | *Olearia colensoi* | 68.0 (± 1.0), 80 (± 100) |
| % Inaka | *Dracophyllum longifolium* | 1.4(± 0.2), 0 (± 80) |
| % Hebe | *Hebe elliptica* | 2.3 (± 0.3), 0 (± 90) |
| % Tētēaweka | *Olearia oporina* | 2.0 (± 0.3), 0 (± 100) |
| *% Senecio reinoldii* | *Senecio reinoldii* | 0.3 (± 0.1), 0 (± 60) |
| % Rātā | *Metrosideros umbellata* | 0.8 (± 0.2), 0 (± 70) |
| *% Coprosma* spp. | | 0.2 (± 0.1), 0 (± 60) |
| *% Pseudopanx* spp. | | 0.3 (± 0.1), 0 (± 80) |
| *% Myrsine chathamica* | | 0.1 (± 0.0), 0 (± 25) |
| % Punga | Tree fern species | 0.4 (± 0.1), 0 (± 100) |
| % other | Other species in the canopy | 1.6 (± 0.3), 0 (± 95) |
| % sky | | 21.5 (± 0.8), 10 (± 100) |
| ***Transects (n = 77)*** | | |
| % Ground Cover | Proportion of the transect covered in dense foliage (up to waist height) | 34.4 (± 4.3), 22.5 (± 100) |
| % Canopy Cover | Proportion of transect covered with some type of vegetation | 71.4 (± 3.5), 80 (± 100) |
| Slope | Slope of transect | 15 (± 1), 15 (± 39) |
| Aspect | Aspect of transect | 191 (± 13), 200 (± 358) |
| Distance to coast | Distance from transect to nearest point on the coast | 94 (± 8), 91 (± 364) |
| Elevation | Elevation of the transect | 46 (± 3), 43 (± 81) |

## Correction factors

The terrain on Taukihepa is variable and often steep, despite the island's high point being only 230 m above sea level. We calculated from our measure of slope on each transect that in some cases this could lead to the underestimation of the actual surface of the island by as much as 15%. We therefore estimated the three-dimensional surface area of each manu by adjusting the two-dimensional areas with the average slope recorded over all transects placed within the manu.

Determining burrow occupancy using a burrowscope is known to underestimate the true number of chicks present according to a decreasing function between probability of detection of the burrow occupant and its distance from the burrow entrance. We therefore adjusted the observed occupancy measures on each manu according to the detection correction function provided by McKechnie *et al.* (2007).

## Manu-wide estimates of chick numbers

The manu-wide estimates of entrance density ($E$) and chick density ($C$) were calculated as weighted means over all sections, the weights being proportional to the areas of each section.

## Harvest estimates

Researchers were provided with the total number of chicks harvested from each *manu* in the year of our field survey there to allow estimation of harvest intensity (proportion of chicks harvested; H). The surveys of these six individual manu were permitted by the families who bird them under the proviso that the resulting information was kept anonymous. A condition of the University of Otago ethics permit given for this research is that the material is reported anonymously. Each family was supplied with a confidential report detailing our findings and copies of each report are also stored at the Department of Zoology, University of Otago. This paper draws together summary information from these reports but cannot list the individual manu involved or any of their detailed results.

If we assume the same harvest intensity for the surveyed and unsurveyed areas, we can estimate the total number of chicks harvested as:

$$H = \frac{H_S}{C_S} C_U + H_S = H_S \left( \frac{C_U}{C_S} + 1 \right)$$

where $C_s$ and $C_u$ are the estimates of the total number of chicks on the surveyed and unsurveyed areas respectively, and $H_s$ is the total number of chicks harvested on the surveyed areas.

## Overall island population estimates

### Vegetation classification

For the purposes of this analysis we classed the vegetation into three distinctive vegetation types recognised by Fineran (1973); (i) Cushion Bog (or *Pakahi*) and low scrub; (ii) Broadleaf Podocarp forest; and (iii) Tūpare (*Olearia colensoi* Hook.f., also known as "Muttonbird scrub") dominated coastal forest. The island is almost completely ringed by a dense canopy of Tūpare where all manu are located and where the bulk of the Sooty Shearwater burrows on the island occur – we classified this entire vegetation type as "birded area". The other canopy species occasionally associated with this vegetation type are listed in table 1. The understory in the birded areas varied from completely open (0% ground cover) to completely closed (100% ground cover). A very sharp fringe zone into the Pakahi typically defines the boundary of this forest type. The Pakahi is a boggy peat habitat typically containing stunted scrub dominated by *Dracophyllum longifolium* R.Br.; Inaka, *Leptospermum scoparium* J.R. et G.Forst.; Mānuka, and *Metrosideros umbellata* Cav.; Rātā, (Fineran 1973). No burrows were ever recorded in this boggy waterlogged habitat. The third main vegetation type, the large Broadleaf/Podocarp forest is almost entirely restricted to one location in the northern lee of a large ridge that crosses the island (fig.1). This entire forest is locally referred to as "Paopuka" and was subject to a full survey in 2005 following the same procedures as described above for the manu surveys.

A number of other small pockets of Broadleaf/Podocarp forests are also scattered around the interior of the island. We termed these areas "bush fragments". Some information on entrance density in these areas was available from visiting a number of them over the years of surveying.

### Mapping

We used a combination of two aerial photographic images taken from surveys flown in 1977 and 1988, ground-truthed with GPS positions, plus our experience on the ground to classify each vegetated area of the island into one of the three main types (fig.1). When classifying these areas we excluded coastal regions of exposed bare rock. Where the vegetation canopy itself appeared to mark the coastline on our aerial images we set the boundary approximately 5 m in from the discernable edge in order to allow for overhangs and cliff edges.

### Relating vegetation and habitat variables to entrance and chick density

We sought to increase the power of our overall estimates of chick density by building separate models to link vegetation descriptors to entrance density estimates for circular plots and for transects. A Principal Components Analysis (PCA) was performed using the habitat variables measured in each circular plot (table 1) on all six manu (n=1 041). Individual scores from the first two PCA axes were then used as predictors of entrance density in a multiple regression. Stepwise regressions using backwards elimination ($\alpha$= 0.1) were performed using entrance and chick density estimates as response variables, and slope, elevation, distance from coast, aspect, %ground cover and %canopy as predictors on all transects (n=77). All analyses were performed in Minitab (v14.1).

### Estimating total number of chicks

Each coastal section of unsurveyed birded area was flanked by two surveyed manu. We therefore estimated the entrance and chick density in each unsurveyed birded area using the weighted mean and standard error of the corresponding estimates from the two neighbouring surveyed manu, where the weights were proportional to the area of these two manu.

The number of entrances located in the "bush fragments" was estimated from 52 circular plots randomly placed within six of these fragments in 2004 and 2005. The overall estimate of occupancy from the Paopuka was used to estimate the number of chicks in these areas.

### Estimating total number of pre-breeders and adults

Our estimate of the total number of chicks present on Taukihepa before any harvesting took place was used to estimate the total number of juveniles, pre-breeders and adults associated with the island, using the following steps:

1. We constructed an age-based population model. The model involved a post-breeding census and contained 21 age-classes. The last age-class contained those individuals that were at least 20 years old. The details of this are given below with details of the other parameters used in the model.
2. We used the model to estimate the stable age distribution for the population.
3. We used the stable age distribution as an estimate of the current age distribution, and therefore the proportion of the population that were chicks.
4. We divided our estimate of the number of chicks by the estimate of the proportion of the population that were chicks to obtain an estimate of the total population size. Using analogous calculations, we estimated the number of breeders, non-breeders, pre-breeders and juveniles.

The parameters in the population model were specified as follows:

(a) Except for the first two years of life, the annual survival rate was assumed to be the same for all age-classes, and was set at 0.952 (95% CI = 0.896-0.979; Clucas *et al.* in press).

(b) The annual survival rate in the first two years of life was set to the value that led to the population growth rate predicted by the model, matching the observed trajectory of the population over the 14 years of our study (Peery *et al.* 2006), as measured by changes in burrow entrance density on the nearby island of Putauhinu, an intensely monitored harvested island (Newman *et al.* 2008). The resulting estimate was 0.525 (95% CI = 0.203 to 0.828).

(c) The annual fertility rate for each age-class was set at

$$f_i = \frac{s_{i-1} a_i p b}{2}$$

where $s_{i-1}$ is the annual survival rate for individuals in age-class $i$–1, $a_i$ is the proportion of individuals in age-class $i$ that are adult, $p$ is the proportion of adults that are breeding (assumed to be the same for all age-classes), and $b$ is the proportion of eggs laid that develop into a fledged chick (also assumed to be the same for all age-classes).

(d) An estimate of the proportion of individuals in age-class i that are adult was obtained in two steps. We first obtained an estimate of the proportion in each age-class that would be at the colony using a method proposed by Spendelow *et al.* (2002). This involved fitting an age-specific model for "probability of first-return" to mark-recapture data obtained from banding chicks at The Snares and Taiaroa

Head. We then combined this model with an estimate of the delay between arrival at the colony and first breeding attempt using data from a long-term study of Short-tailed Shearwater, *Puffinus tenuirostris* (Temminck, 1835), on Fisher Island to obtain an estimate of the proportion of individuals in each age-class that are adult. These final estimates led to an estimate of the mean age at first reproduction of 7.9 years (D. Fletcher *et al.*, unpubl. data).

(e) An estimate of the proportion of adults that are breeding in any one season was obtained using additional data from the study of Short-tailed Shearwater referred to above. Over a period of 41 years (1952–92), the average proportion of adults observed to be breeding was 0.722 (SE = 0.018) (S. Bradley, unpubl. data).

(f) An estimate of the proportion of eggs laid that develop into a fledged chick was obtained by calculating a weighted mean of the estimates for The Snares and Whenua Hou (Newman *et al.* in press). The resulting estimate was 0.722 (95% CI = 0.652 to 0.783).

(g) Confidence intervals for the harvest, number of breeding pairs, adults and total population estimates were obtained by Monte Carlo simulation, with each model parameter being resampled 1000 times from its estimated sampling distribution (Davison & Hinkley 1997).

## RESULTS

### Vegetation on the island

The area of Taukihepa estimated to be covered in vegetation was 797 ha, of which 302 ha (38%) could be classed as birded area and 414 ha (52%) as Pakahi, where no burrows were found. The Paopuka forest occupied a further 61.6 ha (8%) of the island, while bush fragments comprised less than 19.0 ha (2%) (table 2). In total our detailed surveys of the six individual manu covered 63.5 ha or 21% of the total estimated birded area available on the island (table 2, fig.1).

No individual PCA axis could explain much variability in the circular plot vegetation data. The first axis accounted for less than 16% and the second for only 10% of the variability. Multiple regression of these two scores against entrance density also resulted in a poor fit, explaining less than 1% (R-Squared) of the observed variability in entrance density. This suggests at best a very weak (or even no) relationship between the manu vegetation and circular plot entrance density on the manu.

Stepwise regressions revealed few relationships between transect habitat variables and entrance or chick density. The global model containing all predictors of entrance density explained only 14% of the variation. The best fitting model predicting chick density was an even poorer fit, explaining only 5% of the variability.

As we were unable to adequately describe more than 15% of the variation in entrance or chick density using any of the habitat variables measured we discarded all attempts to estimate Tītī density on unsurveyed birded areas based on habitat variables.

### Harvest Intensity

The harvest intensity on the surveyed areas was estimated to be 0.18 (95% CI: 0.16–0.21). Use of Equation 1 gives a total estimate of 137 000 chicks (95% CI: 105 000–173 000) being harvested each year from Taukihepa.

### Overall Population estimates

We estimated that the island contained 1.67 million (95% CI: 1.47–1.88) entrances containing 807 000 (95% CI: 712,000–901 000) chicks prior to harvesting on average over the six years of study. This translates to 1.12 million (95% CI: 0.959–1.28) breeding pairs (at the point of egg laying). We estimate that there are 3.24 million (95% CI: 2.93–3.57) adults and 5.31 million (95% CI: 3.98–6.77) Sooty Shearwaters in total associated with Taukihepa just prior to the breeding season.

## DISCUSSION

### Habitat area

The total area of the vegetated portion of Taukihepa we report here (797 ha) is considerably smaller than any previously published estimate of the island's size (930 ha; Fineran 1973). This discrepancy (of around 15 %) is likely to be due to a combination of our exclusion of any bare rock and coastal fringes and/or the rounding up of less detailed historical estimates of the islands size based on cruder mapping. We

**TABLE 2**
**Estimated area, number of entrances and chicks on Taukihepa**

| Section of Island | Area (ha) | Entrances (± SE) | Chicks (± SE) |
|---|---|---|---|
| Manu: surveyed | 63.5 | 320 000 (± 69,700) | 161 000 (± 80,800) |
| Birded area: unsurveyed | 238 | 979 000 (± 68,900) | 587 000 (± 94,000) |
| Total birded area | 302 | 1 300 000 (± 98,000) | 748 000 (± 117,000) |
| Paopuka | 61.6 | 345 000 (± 28,000) | 55 600 (± 16,000) |
| Bush fragments | 19.0 | 28 000 (± 15,800) | 3 073 (± 3,490) |
| All colonies combined | | 1 670 000 (± 103,000) | 807 000 (± 118 000) |
| Pakahi | 415 | | |
| Total vegetated area | 797 | | |
| Total land area[1] | 930 | | |

[1] From Fineran (1973)

consider our estimate of the vegetated portion of the island containing Sooty Shearwater colonies to be the most robust currently available, as it is based upon extensive ground-based surveying, GPS fixes and detailed aerial images.

The classification of the vegetated portion of the island into the three main vegetation types was a relatively simple procedure, primarily due to the sharply defined and easily discernable boundaries that typically exist between the vegetation types. The results from this classification reinforce the importance of Sooty Shearwaters to this island's ecosystem – with Sooty Shearwater colonies occupying 48.0% of the vegetated area of the island.

## Lack of habitat predictors of burrow density

Our attempts to explain variation in Sooty Shearwater burrow entrance and chick density within the birded area vegetation using any of the habitat variables were largely unsuccessful. This allowed us to adopt a simple approach ignoring habitat when estimating the number of entrances in the remaining unsurveyed birded area. Previous attempts to relate habitat variables to Sooty Shearwater entrance density on other islands have shown mixed results. Some studies have reported differences in entrance densities between broad vegetation types (Charleton 2002, Scott *et al.* 2008, unpubl. data). For example, Charleton (2002) reported that 34.4% of the variation in entrance density on Whenua Hou (Codfish Island) could be explained by habitat and location variables on the eastern side of the island. However, a more recent study of this entire island by Scott *et al.* (2005, unpubl. data) found that only 25.6% of the variation in entrance density could be explained by vegetation and location descriptors when sampling was confined within known colonies. Distance from coast (negative), elevation (positive), canopy height (negative), and the elevation*aspect interaction were found to be weak but significant predictors of entrance density. We also found very weak support for distance from coast and elevation as predictors, but none for aspect, so our results are only partially consistent with earlier studies. Reyes-Arriagada *et al.* (2007) investigated habitat predictors of Sooty Shearwater entrance density on Guafo, an island in Chile. They found no relationship with slope (although they claim the birds preferred steeper slopes between 19–37 degrees), but reported that birds preferred north- and west-facing aspects. We suggest this may have been an artifact of their study design as only a small portion of the northwest region of the island was sampled. In contrast, we found no relationship between entrance density and slope. Furthermore, entrances were found over a wider range of slopes (table 1).

The apparent differences between these studies may result from the islands being so different in size, location, flora and fauna (including introduced species), geology and anthropogenic histories. All of these factors could conceivably mask or confound any common pattern. Nevertheless within the birded area of Taukihepa, entrance density was found to be relatively uniform (0.44 ± 0.11 entrances $m^2$), giving us confidence in our approach.

## Harvest Intensity

Harvest intensity was found to be highly variable on the six manu surveyed on Taukihepa (range 5.00–31.0%). These surveyed manu were not selected randomly because we were able to sample only where birding families welcomed the research. Therefore our extrapolation of the average harvest intensity from these manu to estimate the number harvested on the unsurveyed regions is made with caution.

No relationship between harvest intensity and chick density has been described (Newman *et al.* 2008). Traditional Ecological Knowledge and management by the birders includes manipulation of burrow length and vegetation in ways that are aimed at increasing burrow density (Kitson & Moller 2008, Moller & Lyver in press), in which case burrow entrance density may become higher on birded ground. On the other hand if harvesting decreases overall population abundance, unharvested refuge areas within the birded area may come to have higher abundance than we measured on birded manu.

The only other published record of Sooty Shearwater harvest intensity comes from the adjacent Poutama Island. Lyver (2000) reported harvest intensities of 16.9% (95% CI: 12.6–24.4) and 22.6% (95% CI: 18.4–28.3) on this smaller island (~60 ha) in 1994 and 1995 respectively – comparable intensities to those reported here.

## Reliability of Taukihepa population estimate

Our surveys on Taukihepa were carried out over six years. Breeding burrow entrance density was very stable from one year to the next (Newman *et al.* 2008), whereas burrow occupancy varied greatly between years according to wide variation in breeding success (Wooller *et al.* 1990, Newman *et al.* in press). It is therefore most appropriate to combine our estimates from single years into a single overall Taukihepa population estimate. There is tentative evidence that more crowded colonies have lowered breeding success (Newman *et al.* 2008, Newman *et al.* in press), so we used a combined estimate from The Snares (high density) and Whenua Hou (low density) to extrapolate to Taukihepa. Breeding burrow density is near the middle range on Taukihepa (Newman *et al.* 2008) so our extrapolation from the median is the most appropriate. Repeated monitoring of transects at nearby Putauhinu Island and Whenua Hou island (fig. 1) showed that the Tītī population was approximately stationary over the period of our study (Newman *et al.* 2008). Therefore combining surveys between years is likely to give a robust estimate of the number of breeding attempts (eggs laid). Overall, then, our field sampling and bootstrapping approach will have provided robust estimates of average chick numbers and harvest intensity, and a measure of the uncertainty around those estimates.

Assessing the reliability of the overall Sooty Shearwater population estimate is more problematic because it relies on a population model to estimate the population's age structure. Use of breeder "skipping rate" from Short-tailed Shearwaters is unlikely to have introduced much error because elasticity analyses show that this parameter has relatively little impact on population outcomes (Hunter *et al.* 2000). The key determinant of population growth rate and population structure is the adult survival rate, a parameter that we have been able to estimate accurately by allowing for pre-breeder emigration rates (Clucas *et al.* 2008).

## Future population monitoring and trends

The Sooty Shearwater population on Taukihepa may now increase following recent eradication of Black Rats and reduction in Weka, *Gallirallus australis* (Sparrman, 1786), abundance on Taukihepa (Coote & Blackwell 2006, Dillingham *et al.* 2007). Potential changes resulting from

increased fledging rates of Tītī must be placed within the wider context of other factors that potentially influence Sooty Shearwater populations, including future changes in oceanic productivity, climate change, fisheries bycatch and harvest pressure. Better understanding of density-dependent processes acting within Sooty Shearwater populations is also needed before the future trajectory of the population can be predicted (Newman *et al.* 2008). This study provides a firm baseline for future study of the collective impact of these factors.

## Other petrel species on the island

The only other seabird known to breed on Taukihepa in large numbers is the Korure or Mottled Petrel, *Pterodroma inexpectata* (J.R. Forster, 1844). Its breeding burrows are interspersed with Sooty Shearwater burrows on the steep northeastern face within the Paopuka forest (Scott *et al.* 2005) and the lower gullies around Boat Harbour (S. Bull, pers. comm.). Burrowscoping of all transects on the six manu surveyed did not reveal any Mottled Petrels, so we have assumed that they are not present in the Tūpare classified vegetation type. Our 2005 survey of Paopuka differentiated between Mottled Petrel and Sooty Shearwater burrows and estimated that there were around 21 000 (± 9 310, 95% confidence interval) Mottled Petrel chicks in the Paopuka area alone (Scott *et al.* 2005).

## The importance of seabirds and birding

Burrowing seabirds such as Sooty Shearwater are known to be important components of island ecosystems, often being described as ecosystem "drivers" or "engineers". Although the true extent of their role and influence is only now starting to be understood (Harding *et al.* 2004, Hawke & Holdaway 2005, Hawke & Newman 2005, Bancroft *et al.* 2005, 2008, Fukami *et al.* 2006, McKechnie 2006), clearly the contributions seabirds can make to island ecosystems are considerable. Our estimate of over five million Sooty Shearwaters associated with an island of only 930 ha in size is astounding and provides some insight into the scale of the island–seabird interactions and processes operating there.

Taukihepa represents 58% of the total vegetated area on all Tītī Islands (Newman *et al.* 2008). The entrance and chick densities as well as harvest intensity reported here are therefore likely to be roughly indicative of the national averages on the remainder of the Tītī Islands. If so, an overall national annual harvest of Sooty Shearwater chicks will have been approximately 300 000 chicks on average during our study (1999–2005). This is comparable to the overall take of Short-tailed Shearwater chicks in Tasmania during the 1980s (Skira 1990, 1994), but 70% higher than the average harvest between 1994 and 1997 (I.J. Skira, *in litt.* cited in Moller 1998). The Rakiura harvest intensity of around 18% reported here is half that reported by Skira & Wapstra (1980) for Tasmanian commercially harvested colonies, and a fifth of that recorded at some recreationally harvested colonies. Unlike the Tasmanian Muttonbird harvest, which has declined over the past century (Skira 1990), the Rakiura harvest grew rapidly after the Second World War and has maintained current levels over the past two decades (Newman *et al.* 2008). The supreme importance of the harvest for Rakiura Māori cultural identity, maintenance of knowledge and connection to ancestral lands (Kitson & Moller 2008) motivated this detailed study of chick numbers, population size and harvest intensity on Taukihepa, the main island where harvests occur. The *Kia Mau te Tītī Mo Ake Tōnu Atu* ("Keep the Tītī Forever") research project will now combine this study with surveys on other islands, analysis of trends in harvest success (Lyver *et al.* 1999, Newman *et al.* 2008) and population models incorporating climate change to predict whether the Tītī will remain plentiful enough for the Rakiura *mokopuna* (grandchildren) to continue their cultural inheritance and traditional wildlife harvest.

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(accepted 5 August 2008)

# DECLINE OF SOOTY SHEARWATERS, *PUFFINUS GRISEUS*, ON THE SNARES, NEW ZEALAND

by Darren Scott, Paul Scofield, Christine Hunter and David Fletcher

(with two text-figures and six tables)

**Scott, D., Scofield, P., Hunter, C. & Fletcher, D.** 2008 (31:x): Decline of Sooty Shearwaters, *Puffinus griseus*, on The Snares, New Zealand. *Papers and Proceedings of the Royal Society of Tasmania* 142(1): 185–196. ISSN 0080-4703. Department of Zoology, University of Otago, Box 56, Dunedin, New Zealand (DS); Canterbury Museum, Rolleston Avenue, Christchurch, New Zealand (PS); CNSM/ LAB Biology and Wildlife Department, 211 Irving 1, University of Alaska, Fairbanks, Alaska, 99775, USA (CH); Department of Mathematics and Statistics, University of Otago, Box 56, Dunedin, New Zealand (DF*). *Author for correspondence. Email: dfletcher@maths.otago.ac.nz

The number of Sooty Shearwater, *Puffinus griseus,* burrow entrances on North East Island of The Snares, New Zealand, declined from 3 288 000 (95% CI = 3.1–3.5 million) entrances between 1969 and 1971, to 2 061 000 (95% CI = 1.9–2.2 million) between 1996 and 2001. This represents a decline of 37% over 27 years, or 1.72% per year (95% CI = 1.35%–2.12%). It is not known whether burrow occupancy has also declined. Possible reasons for decline include fisheries bycatch, climate change, and losses on other islands to predation by introduced mammals and harvest of chicks (muttonbirding). Reliable population estimates of other Sooty Shearwater populations will be important for establishing baseline estimates against which future population trends can be evaluated. If the decline in burrow numbers represents a similar decline in Sooty Shearwater numbers on The Snares and elsewhere, the decline reported here is substantial and enough to warrant listing of the Sooty Shearwater as a "vulnerable" species, according to criterion A3 designated by the International Union for Conservaion of Nature and Natural Resources (IUCN) in 2001.

**Key Words: Sooty Shearwater, *Puffinus griseus,* The Snares, population decline, fisheries bycatch, climate change, predation, harvest.**

## INTRODUCTION

Marine birds are sensitive to perturbations in climate, prey abundance and food-web structure. In recent decades increasing pressure has been put on the world's oceans from fisheries and global environmental changes associated with a burgeoning human population (Worm *et al.* 2006). Seabirds are increasingly being used as sensitive indicators of the state of the world's oceans (Furness & Camphuysen 1997). Seabird population decline is widespread, with 50% of taxa in the largest group of marine birds, the Procellariiformes, listed by the IUCN as threatened, vulnerable or endangered (Collar *et al.* 1994).

Procellariiformes are long-lived, monogamous species with low reproductive output and delayed maturation (Warham 1990, 1996). Their populations are particularly sensitive to any fluctuation in adult survival (Danchin *et al.* 1995, Hamilton & Moller 1995, Russell 1999). Demographic consequences of a significant mortality of breeding birds can therefore be rapid and severe, and recovery slow, e.g., the near extinction of the Black Petrel *Procellaria parkinsoni* G.R. Gray, 1862, on Little Barrier Island (Imber 1987). Maturation is delayed, so any change in fledging or juvenile survival may take a generation or more to be detectable in population trends (Fletcher *et al.* unpubl. data).

Sooty Shearwater (Tītī), *Puffinus griseus* (J.F. Gmelin, 1789), is a highly abundant trans-equatorial migrant seabird that plays a major role in the ecology of the islands of southern New Zealand and South America, where it breeds (Shaffer *et al.* 2006). The current total New Zealand population size is estimated to be 19–23 million individuals (Newman *et al.* unpubl. data). Chicks ("muttonbirds") are harvested by Rakiura *Māori* (indigenous people) (see Kitson & Moller 2008). Approximately 360 000 chicks have been harvested annually in recent years (Newman *et al.* 2008). However, of indicators of Sooty Shearwater decline has been found. Harvest rates of Sooty Shearwater chicks on Poutama Island decreased by 42–47%, or 3.6% to 3.9% per year, between 1989 and 1998 (Lyver *et al.* 1999). Furthermore, a 64.4% decline was found in the number of adult Sooty Shearwaters found dead on beaches in New Zealand between 1961 and 1999, or 1.3% per year (95% CI = 0.3% to 1.7% per annum) (Scofield & Christie 2002). Declines have also been recorded elsewhere; for example Sooty Shearwater numbers in the Californian current system declined by 90%, or 7.4% per year, between 1987 and 1995 (Veit *et al.* 1997).

Driftnet fishing for squid and salmon occurred in the North Pacific from 1978 until the end of 1992 (Uhlmann 2002, 2003, Uhlmann *et al.* 2005). Estimates of the impact of these fisheries on marine ecosystems vary. Sooty Shearwater and Short-tailed Shearwater, *Puffinus tenuirostris* (Temminck, 1835), were the largest component of the bycatch, both numerically and in biomass. Shearwaters may still be a major component of the bycatch in salmon fisheries in the North Pacific (Artyukhin & Burkanov 2000). Ogi *et al.* (1993) predicted a population growth rate ($\lambda$) of between 0.993 and 0.972 for the breeding stock of Sooty Shearwaters during the period driftnet fisheries operated. However, this estimate has not been confirmed by actual estimates of population trends at their breeding colonies (Warham 1996) and all the inputs to Ogi *et al.*'s (1993) model were from Short-tailed Shearwater.

The Snares (48°01'S, 166°36'E) is a group of small islands lying 105 km south-southwest of Stewart Island, New Zealand. Surveys conducted between 1969 and 1971 estimated that collectively these islands had a population of 2.75 million breeding pairs of Sooty Shearwater (Warham & Wilson 1982). Our study repeats the counts undertaken by Warham & Wilson (1982) on North East Island, the largest of The Snares islands. Burrow counts previously undertaken in 1986/87 at 30 permanently marked vegetation plots are

also repeated in this study. Our primary aim is to determine whether a measurable decline in the Sooty Shearwater breeding population has occurred in the decades since the original surveys were undertaken. We also estimate the generation time of Sooty Shearwaters from recently available population parameters and ask whether any decline observed on The Snares has reached the critical threshold that would lead to the species being listed as threatened according to IUCN (2001) criteria.

## MATERIALS AND METHODS

### Study site

North East Island, also referred to elsewhere as Main Island, is 238.1 ha in area. The principal habitat types on the island are *Olearia lyallii* Hook.f. (tree daisy) forest, and maritime tussock grassland dominated by *Poa astonii* Petrie and *P. tennantiana* Petrie (Warham & Wilson 1982, Hay *et al.* 2004).

### Burrow entrance counts

We define a burrow entrance as a single entrance leading to one or more tunnels (following Warham & Wilson 1982). Entrances may have more than one occupied tunnel and individual nests frequently have more than one entrance (McKechnie *et al.* 2007, Hamilton 2000). Assuming that the probability of detecting entrances in different areas and times does not change, this will not affect interpretation of relative trends in shearwater abundance.

There are about 10 000 breeding pairs of Mottled Petrels, *Pterodroma inexpectata* (J.R. Forster, 1844), on The Snares (Warham *et al.* 1977). Mottled Petrel and Sooty Shearwater burrow entrances can be difficult to tell apart. Warham & Wilson (1982) do not state how they separated Mottled Petrel and Sooty Shearwater burrows in their study; however we found that Mottled Petrel burrows were always in dry, open, well-drained sites. Additionally, no Mottled Petrel burrows were found in soil with a depth of greater than c. 50 cm, and tunnels were rarely longer than an arm's length. Furthermore, burrow entrances of Mottled Petrels were smaller than those of Sooty Shearwater (Scott 2005, Scott *et al.* 2006), and were almost always in areas where *Poa* tussock was prevalent; only one occupied Mottled Petrel burrow was found more than 5 m from the *Poa* tussock strip that encircles the island.

We assumed that all burrow entrances with a maximum greatest entrance width of less than 7 cm were Common Diving-Petrel, *Pelecanoides urinatrix* (J.F. Gmelin, 1789). An estimate of the number of diving petrel burrows recorded during these surveys is given in Miskelly *et al.* (2001).

### 1969–71 surveys

Warham & Wilson (1982) sampled 51 quadrats, each 10 x 10 m and randomly placed off marked tracks, between 1969 and 1971. They assessed habitat type and estimated planar habitat size from a map produced in 1971 from aerial photographs (Hay *et al.* 2004). Random number tables were used to obtain distances along and off the track for quadrat placement. The approximate position of these tracks is given in Figure 1 of Warham (1967); however the locations of the quadrats were not permanently marked.

### Permanent vegetation quadrats, 1986/87

The University of Canterbury established 30 10 x 10 m permanently marked vegetation quadrats in the summer of 1986/87. Only 23 quadrats contained recognisable burrows that were occupied. The primary purpose of these quadrats was to map vegetation change, but burrow entrance positions were plotted and the total numbers of entrances was counted.

### Burrow entrance surveys, 1996–2001

In all 357 sampling units, with a combined area of 2.25 ha, were surveyed for burrow entrance density during the late incubation period between 1996 and 2001. Four methods of plot establishment and size were used to address related research questions, described elsewhere (Scofield 2000, Charleton 2002).

#### Quadrats positioned from tracks

Random numbers were used to select a distance along and off the track system for placement of 10 x 10 m plots in Dec–Jan 1996 (n = 60).

#### Transects positioned from tracks

In Dec–Jan 1996, random numbers were used to select a distance along and off the track system for placement of 2-m-wide transects. Transects followed a randomly-assigned direction until 20 used burrow entrances were included and transect length was then recorded to provide an estimate of entrance density (n = 45).

#### Quadrats positioned along transects across the island

Three randomly-placed line transects were chosen, running approximately north-south, east-west and southeast-northwest across the entire island (fig. 1). Plots measuring 10 x 10 m were placed every 100 paces along the transects in January 2000 (n = 101).

#### Circular plots along transects near coast

Random numbers were used to obtain start points for 18 transects (nine each on the east and west coast) that ran perpendicular to the coast. Circular plots of 3 m radius were placed at intervals of 3, 15, 27, 54, 76, 113 and 174 paces from the coast in January 2001 (n = 151).

In addition to the 357 sampling units above, the Sooty Shearwater burrow entrances within the 23 University of Canterbury permanent vegetation quadrats recorded with burrow entrances in 1986/87 were counted in December 1998.

The locations of all sampling units on North East Island are shown in figure 1. We assessed vegetation type, whether the ground cover was open or closed and recorded the number of Sooty Shearwater entrances that were open, and the number of Common Diving-Petrel and Mottled Petrel entrances in each sampling unit. The position of each sampling unit was recorded using a Trimble Geo-explorer II ® differential GPS unit and mapped in Geographical Information System MapInfo 6.0™, on a base map supplied by Land Information New Zealand. Vegetation boundaries and types were digitised from a map produced from aerial photographs taken in 1971 (Hay *et al.* 2004). This allowed the land area under each vegetation type to be calculated and mapped. During the 2000 and 2001 surveys, this map was "ground truthed" in the field to determine changes in

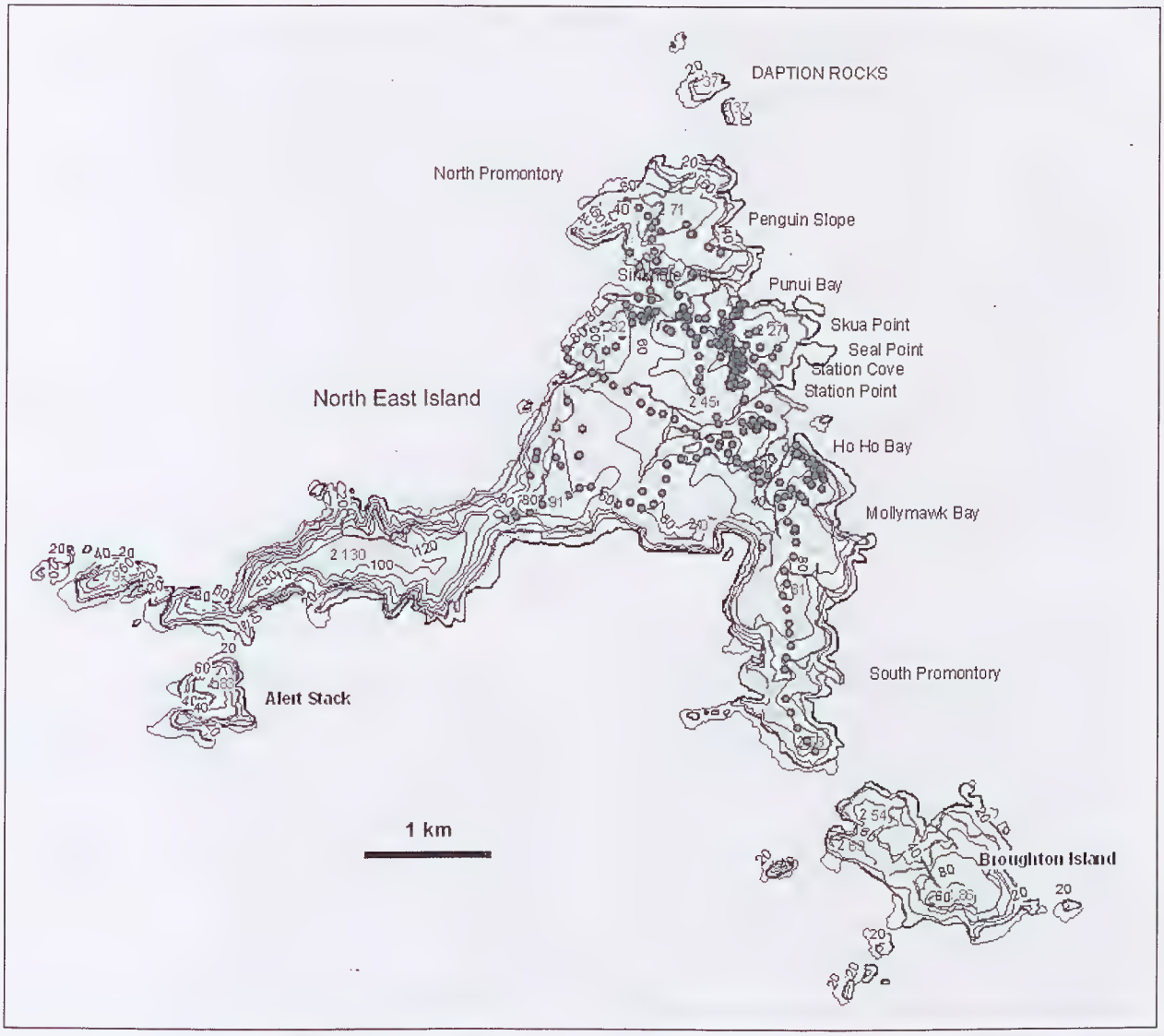


*FIG. 1 — North East Island, The Snares, with locations of sampling units (circles) surveyed for Sooty Shearwater burrow entrances 1996–2001.*

vegetation cover since 1971. Changes in vegetation cover were found to be minor, with a small decrease in the area classified as *Olearia* forest and increase in the area classified as *Hebe elliptica* shrubland (table 1). The estimated area of the island available for Sooty Shearwater burrows excluded the area occupied by penguin colonies. A survey of Snares Crested Penguin, *Eudyptes robustus* Oliver, 1953, detailing colony sizes and locations undertaken in October 2000 (D. Houston, unpubl. data) was used to exclude these areas from the estimated burrowable area of the island.

## Burrow occupancy

The proportion of Sooty Shearwater burrows in which breeding was evident (egg or chick) is referred here as the "occupancy rate". This has been assessed in three general and five specific ways throughout the studies compared here.

### 1. Contents of collapsed nesting chambers

Whilst carrying out fieldwork on The Snares it is difficult to avoid accidentally collapsing burrows that are often situated in fragile soils. Warham & Wilson accidentally collapsed 54 nesting chambers between 3 December 1970 and 24 February 1971. During our study, 86 burrow chambers were collapsed between 26 December 1999 and 20 January 2000; and 62 chambers between 10 January and 5 February 2001. All burrows were carefully reconstructed using available materials, but first the contents (egg or chick) were determined.

### 2. Burrowscoping

A burrowscope consists of a miniature surveillance camera on the end of a long tube that projects an image of the nesting chamber onto a monitor on the surface (Lyver *et al.* 1998). We used burrowscopes to estimate burrow occupancy on the 45 2-m-wide transects (900 burrow entrances) positioned from tracks between December–April 1996 and 1998.

### 3. Measures of passage into and out of burrow entrances in January

Comparisons are possible between two methods: (i) Horning (1976) and Warham & Wilson (1982) estimated occupancy by observing the disturbance of fine wires placed across the 31 burrow entrances between 4 and 22 January 1972. (ii) We used an electronic burrow monitoring system (based

**TABLE 1**
**Sooty Shearwater burrows by vegetation type on North East Island, The Snares**

| Vegetation type | Cover[1] (%) | | Burrow density[2] ($m^{-2}$) | | Area (ha) | | Est. no. of burrows (thousands) | | S.E. (thousands) | |
|---|---|---|---|---|---|---|---|---|---|---|
| Period | 1[3] | 2[4] | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 |
| *Olearia* forest[5] | 61.7 | 55.9 | 1.16 | 0.71 | 146.8 | 133.0 | 1704 | 948 | 104 | 29 |
| *Poa* tussock | 32.1 | 32.5 | 1.95 | 1.23 | 76.5 | 77.3 | 1490 | 950 | 44 | 26 |
| *Brachyglottis*[6] forest | 1.4 | 1.9 | 0.46 | 0.70 | 3.3 | 4.6 | 15 | 32 | 3.8 | 4.1 |
| *Hebe elliptica* | 4.8 | 9.7 | 0.68 | 0.57 | 11.5 | 23.2 | 78 | 131 | 1.7 | 15 |
| Total | – | – | – | – | 238.1 | 238.1 | 3288 | 2061 | 113 | 42 |

[1] Cover is the percentage of the island covered by that vegetation type.
[2] Burrow density is the estimated mean density of burrow entrances.
[3] Period 1 data were collected between 1969 and 1971 (from Warham & Wilson 1982).
[4] Period 2 data were collected between 1996 and 2001.
[5] Includes vegetation from areas in which *Olearia* was present with other vegetation.
[6] Formerly *Senecio*.

on an off-the-shelf burglar alarm system) to record the date and time of entrance and exit events from manual triggering of switches placed in the burrow entrance (Moller *et al.* 2003a). One panel could monitor 16 separate entrances and up to four panels were used simultaneously during this study. Thirty-two of these automated entrance monitors operated at two study sites near Punui Bay between 10–22 January 1999 and 4–22 January 2000.

4. Measures of passage into and out of entrances in April

(i) Horning (1976) and Warham & Wilson (1982) estimated occupancy of 100 entrances chosen at random under *Olearia* forest near Punui Bay in April 1970. In early April, they determined occupancy by feeling for an occupant by hand or by using a long wire.
(ii) In late April, the knocking down of thin wires placed in the burrow entrance, evidence of fresh digging, presence of nestling down in the tunnel, and observation of birds entering these 100 entrances were all used to assess occupancy.
(iii) We used the electronic entrance monitoring system to quantify use of 15 occupied and 17 unoccupied burrows between 1–30 April 1998; and 11 occupied and 21 unoccupied burrows between 1–30 April 1999.

## Surface egg density

Some Sooty Shearwater eggs are laid on the ground outside burrow entrances ("surface eggs"). These can be easily distinguished from eggs which have been ejected from burrows, as the latter are generally dirty and stained by incubation and rolling. Warham *et al.* (1982) estimated the number of surface eggs at two sites of differing burrow entrance density near Punui Bay throughout the 1970 egg laying period. We searched six sites in 1998, varying in size from 100–1600 $m^2$, which were randomly located within *Olearia* habitat type and in reasonable proximity to the research hut in the Boat Harbour. One of the six sites was excavated for another study in 1996 (Hamilton 2000).

## Statistical methods

We used single factor ANOVA to compare the burrow entrance density estimates obtained using four sampling methods (quadrats, transects, quadrats from line transects and circular plots) in the *Olearia* vegetation type. We used stratified sampling methods (Cochran 1977) to estimate the standard error associated with the estimate of the total number of burrows in the different time periods, the strata being the different vegetation types. All analyses were carried out in a Microsoft Excel spreadsheet.

## Generation time

One of the IUCN (2001) criteria used to determine whether a species is classed as "vulnerable" is a decline of ≥50% over the last 10 years or three generations (criterion A1). We estimated the generation time ($T$) of Sooty Shearwater using a standard formula, based on the assumption that all breeding adults have the same reproductive rate and the same survival rate, i.e.,

$$T = \alpha + [s/(\lambda - s)]$$

where $\alpha$ is the age at first breeding (assumed the same for all birds), $s$ is the breeder survival rate, and $\lambda$ is the population growth rate (Gaillard *et al.* 2005). There are no published data on the age at first breeding for Sooty Shearwater, so we used an estimate for a closely-related species, Short-tailed Shearwater (Bradley *et al.* 1989).

# RESULTS

## Burrow entrance counts

During the survey period 1969–71, the density of Sooty Shearwater burrow entrances in different vegetation types ranged between 0.46 and 1.95 entrances per $m^2$ (Warham & Wilson 1982, table 1). Using aerial photographs and a vegetation map, Warham & Wilson (1982) estimated the total planar area of North East Island to be 280 ha. Excluding

areas devoid of soil, they estimated the burrowable area of the island to be 238.1 ha. The area occupied by Snares Crested Penguins was not considered by Warham & Wilson (1982) (K-J. Wilson pers. comm.). The October 2000 survey of Snares Crested Penguins counted 25 851 nests. Each penguin nest occupied approximately 1 $m^2$ (D. Houston pers. comm.), so penguins make c. 2.6 ha unavailable to Sooty Shearwaters.

Three of the methods used to survey burrow entrances in 1996–2001 were not specifically stratified to sample representative vegetation types. Burrow density from all four methods (quadrats, transects, quadrats from line transects and circular plots) could therefore only be compared in the *Olearia* vegetation type. There was evidence for a difference in the burrow density estimates found using the four counting methods (F= 4.66, with 3 and 168 d.f.; P = 0.004), but the differences were small (table 2). Thus, we combined the burrow entrance density estimates from the four methods to generate the most accurate estimate of burrow density possible for 1996–2001. The density of burrow entrances in different vegetation types ranged from 0.57 to 1.23 entrances per $m^2$ (table 1). The estimated total number of Sooty Shearwater burrow entrances on North East Island during our study between 1996 and 2001 is 2.06 million (95% CI = 1.9 to 2.2 million) (table 1).

## Changes in burrow entrance density

We estimate a decrease of approximately 1.23 million Sooty Shearwater burrow entrances (95% CI = 0.95–1.51 million) has occurred between 1969/1971 and 1996/2001. This represents a decline of 37% over 27 years, or 1.72% per year (95% CI = 1.35–2.12%).

Burrow entrance density for the 23 permanent vegetation quadrats that contained burrow entrances in 1986 decreased from 1.34 entrances per $m^2$ (SE = 0.23) to 1.09 entrances per $m^2$ (SE = 0.20) between 1986 and 1998, respectively. The 95% CI for the decrease was 0.05–0.45 entrances per $m^2$. This represents a decline of 19% over 12 years, equivalent to an annual decline of 1.71% per year (95% CI = 0.51–3.09%).

## Conservation status of Sooty Shearwater

Bradley *et al.* (1989) report an estimate of age at first breeding for Short-Tailed Shearwater of 7.0 years (SE = 1.1). They also provide an estimate of adult survival rate of 0.92 (SE = 0.01). The only published estimate of breeder survival rate for Sooty Shearwater is 0.87 (95% CI: 0.80–0.94), given by Scofield *et al.* (2001).

If we use the estimates of the age at first breeding ($\alpha$) and breeder survival rate ($s$) from Bradley *et al.* (1989), together with the estimate of $\lambda$ = 0.983 (95% CI: 0.979–0.987) implied by a 1.7% decline in burrow density, we obtain an estimate of the generation time ($T$) of 21.6 years (95% CI: 17.2–29.3). We used parametric bootstrapping, with 10 000 replicates, to determine the confidence interval for $T$ and for the decline over three generations (Davison & Hinkley 1997). On this basis, we estimate that the observed 1.7% annual decline of Sooty Shearwater burrow entrances equates to a 67% decline over three generations (95% CI: 54–81%). Using the estimate of adult survival for Sooty Shearwater from Scofield *et al.* (2001), we get a value for $T$ of 14.7 years (95% CI: 10.7–28.5), which leads to an estimated 53% decline over three generations (95% CI: 40–78%).

**TABLE 2**
**Sooty Shearwater burrow entrance density on North East Island, The Snares during 1996-2001, for sampling units classified as having the *Olearia* vegetation type**

| Sampling unit[1] | Burrow density[2] ($m^{-2}$) | S.E. | n |
|---|---|---|---|
| Quadrats | 1.15 | 0.11 | 6 |
| Transects | 1.05 | 0.05 | 25 |
| Quadrats from line transects | 1.16 | 0.04 | 38 |
| Circular plots | 0.98 | 0.03 | 103 |

[1] Four types of sampling units were used.
[2] Burrow Density is the estimated mean density of burrow entrances.
Does not include observations from areas in which *Olearia* was present with other vegetation (cf. table 1).

## Burrow occupancy

There is no evidence of a decline in the occupancy of nest chambers that were accidentally collapsed between 1971 and 1999/2000 (table 3; greatest difference $\chi^2$ = 1.66; d.f. = 1; P = 0.20).

Similarly, there was little difference in the frequency of entrance events in January of 1972, 1999 and 2000 ($\lambda^2$ = 1.359; d.f. = 1; P = 0.244). Entrances containing an egg or chick were entered an average of 59.7% of nights, whereas those without eggs or chicks were entered 79.2% of nights (t = 2.055, d.f. = 26, P < 0.05; fig. 2). Entrances in which breeding had failed, or in which an egg had never been laid, had very frequent visits, whilst those that contained a chick were less frequently visited. We conclude that frequency of entrance events in January cannot reliably indicate occupancy, so comparison between 1972 and recent estimates of activity cannot indicate changes in burrow occupancy.

Eighty-six per cent of entrances investigated in April 1972 had some indication of use that Horning (1976) and Warham &Wilson (1982) interpreted as indicative of occupation by a chick. Between 1 and 30 April 1999, 36.1% (s.d.=10.0%,

**TABLE 3**
**Occupancy rate (%) of burrows accidentally collapsed on North East Island, The Snares**

| | 1971[1] (n=54) | 1999 (n=86) | 2000 (n=62) |
|---|---|---|---|
| Adult on egg | 55.6 | 45.3 | 45 |
| Adult, no egg | - | 19.8 | 8 |
| Two adults, no egg | - | 1.2 | 3 |
| One or two adults, no egg | 18.5 | - | - |
| Abandoned egg | 0.0 | 1.2 | 6 |
| Occupancy rate | 74.1 | 67.5 | 61.3 |
| Mottled Petrel[2] | 0.0 | 1.2 | 0 |

[1] Data for 1971 are from Warham & Wilson (1982).
[2] Data are for Sooty Shearwaters unless otherwise indicated.

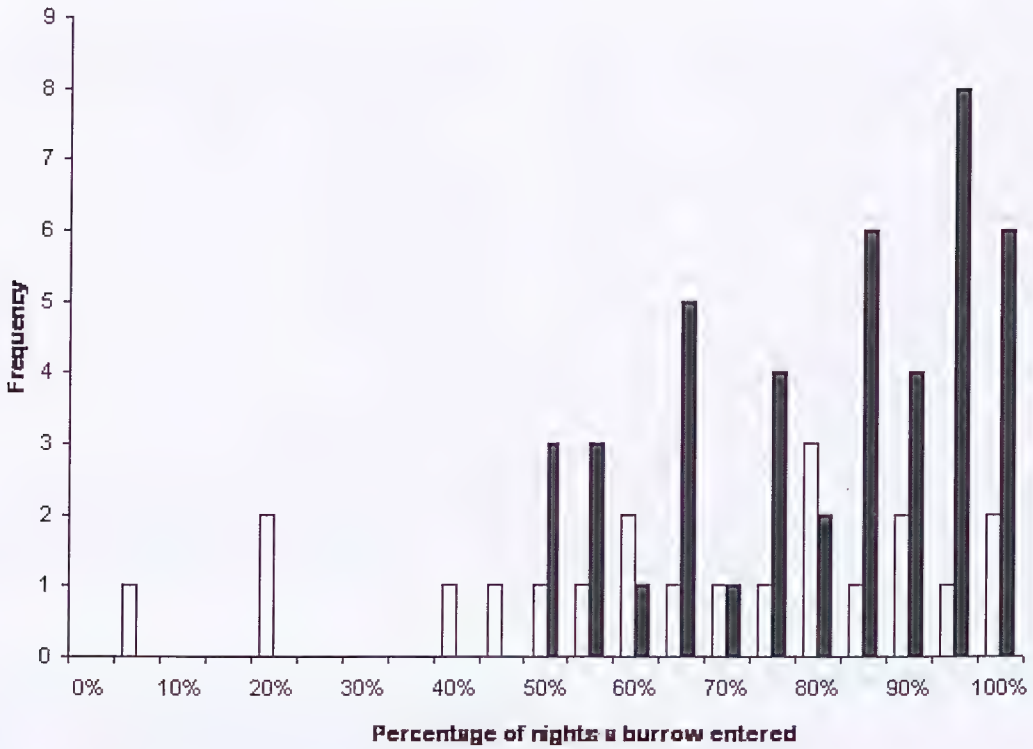


*FIG. 2 — Frequency of Sooty Shearwater burrow entrance events per night, North East Island, The Snares. Dark bars represent burrows without chicks, open bars burrows with chicks. Sixty-four burrows were monitored over 19 nights; 4–22 January 2000 (From Moller* et al. *2003a).*

range 18.8–56.3%) of 32 entrances monitored each night were active. Overall 94% of 32 burrows were visited at least once. Fifteen of these entrances contained chicks or eggs in the 1999/2000 season and 11 did so in April 1999. Non-breeders were absent from the colony during this period (Richdale 1954, R.P. Scofield unpublished. data). It would appear therefore that the true level of occupancy in 1972 may have been closer to half that reported. Explanations for overestimation of occupancy in 1972 include detection of wandering chicks or breeding adults which have holed up overnight in empty burrows after they have fed their chicks. The latter behaviour has been observed in Sooty Shearwaters elsewhere (G. Taylor pers. comm). Given this result, it is significant that only 42 entrances (51% of the 82 thought to be occupied) had direct evidence that a burrow was occupied (a chick was felt by hand or with a wire) in 1972. There was no evidence that the occupancy estimate in 1972 was different from the average burrowscope estimates of occupancy in the late chick phase in 1996–2000 ($t = -1.685$; d.f = 8, $P = 0.13$; tables 4, 5).

Methodological variations make interpretation of the above results extremely difficult. Furthermore, published estimates of occupancy on The Snares using burrowscopes with similar operators and methods indicate high variability between years and sites (table 4, Hunter 2001). Therefore, it is not possible to infer changes over time in average burrow occupancy from the data available.

### Surface egg density

There was no evidence of a decline in surface egg density between 1970 and 1998 ($t = 0.015$; d.f. = 6; $P > 0.89$; table 6). Similarly, there was no evidence that the mean number of surface eggs per burrow entrance changed ($t = 0.0.32$; d.f. = 6; $P > 0.76$; table 6).

## DISCUSSION

### Burrow occupancy

Our monitoring of entrances of occupied and empty burrows using an electronic system showed that estimates of traffic in and out of entrances is not a reliable indicator of occupancy by breeding Sooty Shearwaters, especially in the middle of the breeding season. Hamilton (1998) reached a similar conclusion from her mainland New Zealand studies using stick barricades at burrow entrances. Entrances that were visited frequently are more likely to contain failed breeders or prospecting birds than breeders (fig. 2). The supposition that burrows with activity at the entrance in April contained chicks (Horning 1976, Warham & Wilson 1982) was based on the assumption that chicks in the period from first emergence to fledging are faithful to their natal burrow. Traditional Ecological Knowledge of the Rakiura muttonbirders (Charleton 2002) and radio-tracking (C.M. Hunter unpubl. data) showed that chicks wander and explore other burrows from mid- to late April. Entrances in areas close to "take-off points" such as Punui Bay on North East Island, where one of our automatic monitoring plots was concentrated, are likely to have the highest level of unoccupied burrow visitation.

There is the potential for bias in the estimation of burrow occupancy rates from the inspection of collapsed burrows. Burrows close to the surface are more likely to collapse. If these burrows are favoured for breeding, then occupancy

**TABLE 4**
**Occupancy rate (%) of Sooty Shearwater burrows on North East Island, The Snares, from burrowscope data, by breeding stage and vegetation type (from Hunter 2001)**

| Year | Breeding stage | Estimated burrow occupancy rate (%) | | |
|---|---|---|---|---|
| | | *Poa* tussock | *Olearia* forest | Fern |
| 1996 | Incubation | 42.2 | 65.9 | 50.5 |
| 1997 | Incubation | 11.1 | 28.8 | 17.1 |
| 1998 | Incubation | 25.9 | 34.3 | 36.7 |
| 1996 | Early chick | 30.7 | 48.3 | 34.9 |
| 1997 | Early chick | 5.6 | 12.8 | 6.0 |
| 1996 | Late chick | 43.8 | 48.1 | 33.2 |
| 1997 | Late chick | 10.8 | 11.6 | 1.0 |
| 1998 | Late chick | 21.9 | 22.9 | 16.2 |

**TABLE 5**
**Indications of Sooty Shearwater occupancy in 100 monitored burrows on The Snares, in April 1972 (from Horning 1976)**

| | Only indication of occupancy | Additional indication of occupancy | | |
|---|---|---|---|---|
| | | Down | Bird felt with wire loop | Digging |
| No indication of occupancy = 14 | | | | |
| Bird felt with wire loop | 36 | 1 | – | 1 |
| Bitten hand | 6 | – | – | – |
| Wire fence disturbed | 15 | 13 | 3 | – |
| Fresh digging | 6 | – | – | – |
| Down in burrow mouth | 4 | – | – | – |
| Bird seen entering burrow | 1 | – | – | – |

**TABLE 6**
**Density of Sooty Shearwater eggs laid on the ground outside burrow entrances ("surface eggs") at sites on North East Island, The Snares, in 1970 and 1998**

| Site | Area ($m^2$) | Burrow entrance density ($m^{-2}$) | Number of surface eggs | Surface egg density ($m^{-2}$) | Surface eggs per burrow entrance |
|---|---|---|---|---|---|
| *1998* | | | | | |
| A | 600 | 0.52 | 8 | 0.013 | 0.026 |
| B | 400 | 1 | 9 | 0.023 | 0.023 |
| C[1] | 100 | 0.91 | 4 | 0.040 | 0.044 |
| D | 650 | 0.61 | 11 | 0.017 | 0.028 |
| Low burrow density | 600 | 0.81 | 7 | 0.012 | 0.014 |
| High burrow density | 1600 | 1.1 | 20 | 0.013 | 0.011 |
| *1970*[2] | | | | | |
| Low burrow density | 3160 | 0.73 | 58 | 0.018 | 0.025 |
| High burrow density | 2860 | 1.06 | 55 | 0.019 | 0.018 |

[1] Disturbed site (Hamilton *et al.*1998).
[2] Data for 1970 are from Warham *et al.* (1982).

rates may be overestimated. Conversely, if these burrows are less favoured, then occupancy could be underestimated. Additionally, areas with greater breeding success may have more burrows closer to the surface. Estimates of burrow occupancy from detection of chicks when felt by hand or with the use of a wire also have limited accuracy. A comparison gave no evidence for a difference between Warham & Wilson's (1982) and our burrow occupancy estimates. However, these comparisons will have had low statistical power because of the large variation in occupancy between years (table 4 and McKechnie *et al.* 2007) which results from large inter-annual variation in the proportion of adults laying eggs and breeding success (Newman *et al.* 2008, Newman *et al.* in press). Therefore we have no evidence that there has been decline or increase in burrow occupancy.

Density of breeding burrow entrances provides a stable index of breeding density that smoothes out inter-annual variation in burrow occupancy (Newman *et al.* 2008). There is also a close relationship between absolute chick density (as estimated by the burrowscope) and burrow entrance density on different islands and within family muttonbirding territories (Newman *et al.* 2008). Therefore it is most likely that changes in burrow entrance density that we observed on North East Island reflect a commensurate change in actual density of Sooty Shearwater breeding pairs during the interval between our studies.

## Decline of Sooty Shearwaters

Our study confirms that the number of Sooty Shearwater burrow entrances on North East Island, The Snares, has declined markedly in the past 30 years. Although there was no evidence that the number of surface eggs has also declined, results for this measure are inconclusive because of large variation in the number of surface eggs between areas and the small number of areas and years sampled. The Traditional Ecological Knowledge of the Rakiura muttonbirders records that the number of surface eggs fluctuates markedly between years (H. Moller unpubl. data), so a much longer run of observations will be needed to determine whether or not the number of surface eggs has changed.

There is strong evidence that the decline documented here is not restricted to The Snares islands and that the causes of the decline may also be affecting other populations of Sooty Shearwaters and other shearwater species. A 90% decline in Sooty Shearwater numbers observed in the Californian current system over the period 1987–1994 (Veit *et al.* 1997) has been attributed to water temperature variations and climate change, but changes in bird distribution within the counting zone may have contributed (Oedekoven *et al.* 2001). Declines in Sooty Shearwater populations documented on the New Zealand mainland (Hamilton *et al.* 1997, Jones 2000) have been attributed primarily to predation by introduced small mammals. Various islands in the Hauraki Gulf of the North Island of New Zealand held small populations of Sooty Shearwaters until at least the 1960s (Falla 1934 and Department of Conservation banding records) that have now gone (R. Pierce, A. Booth and G. Taylor pers. comm., R.P. Scofield unpubl. data). An estimated 80 breeding pairs were recorded on Motunau Island, North Canterbury, in 1962 (Taylor 1967) although Scofield (2000 unpubl. data) found this population to be virtually extinct. It is significant that some of these declines or local extinctions have occurred in the absence of introduced predators or harvest, as observed in this study on The Snares and recently on Whenua Hou (Lyver *et al.* 1999). Recent declines in closely-related species, incuding Short-tailed Shearwater (Walsh *et al.* 1997), Fleshy-footed Shearwater, *Puffinus carneipes* (Gould, 1844), (Scofield 2000) and Pink-footed Shearwater, *Puffinus creatopus* (Coues, 1864), (Ainley *et al.* 1995), have also been documented.

## Possible reasons for the Sooty Shearwater decline

It is impossible to isolate a single cause for the decline in Sooty Shearwaters. Although the North Pacific squid fishery ceased in 1992 (Uhlmann *et al.* 2005), salmon gill net fisheries have continued throughout the North Pacific and these fisheries still take large numbers of shearwaters (Artyukhin & Burkanov 2000). Approximately 40% of shearwaters killed in the Japanese squid driftnet fishery were of breeding age (Ogi *et al.* 1991a, b). Shearwaters are long-lived, monogamous species with a low reproductive output (Warham 1990, 1996). They take one or more seasons to develop a pair bond and only ever lay one egg per season (Richdale 1963, Skira 1991). Sensitivity analyses of shearwater demography have shown that population growth rate is most sensitive to any change in the rate of mortality of breeding age birds (Hamilton & Moller 1995, Hunter *et al.* 2000). Ryan & Boix-Hinzen (1999) found a consistent male bias in fisheries bycatch for a number of petrel species taken off South Africa. Sooty Shearwaters are killed in New Zealand trawl fisheries and although the full extent of this mortality has not been quantified, it is known that the proportion of killed birds which are male is high (Robertson 2000). For example, 83% (95% binomial confidence intervals 72.2–94.2%) of a spatially – and temporally – representative sample of Sooty Shearwaters (n=205) taken mostly in the New Zealand squid trawl fishery were males (C.J.R. Robertson and R.P. Scofield, unpubl. data). If this sex-biased mortality was due to competitive exclusion, a similar bias would be expected in the North Pacific bycatch; however the available data on the sex ratio of shearwaters killed in the North Pacific are not consistent (Gould *et al.* 2000).

Climate fluctuations such as the El Nino/La Nina Southern Oscillation and the Pacific Decadal Oscillation trigger short-term and medium-term shifts in wind and currents (Mantua *et al.* 1997, Salinger & Mullan 1999) that impact on bird productivity and potentially survival, movements and distribution (Veit *et al.* 1997, Lyver *et al.* 1999, Shaffer *et al.* 2006). Diet sampling and tracking studies show that Sooty Shearwaters forage at the highly productive upwelling region at the Sub-Antarctic Front (SAF), the boundary between the cool, sub-Antarctic waters and the warmer sub-tropic waters (Weimerskirch 1998, Shaffer *et al.* 2006). It is believed that inter-annual and long-term increases in global heat mass may push the SAF southwards. Sea surface temperature (SST) has been increasing in the waters between 45°S and 20°S since 1961 (Hurrell & Trenberth 1999) and average SST and chlorophyll *a* measured near Marion Island has increased (Hunt *et al.* 2001), probably as a result of the SAF moving southward. Decline in the number of Rockhopper Penguins, *Eudyptes chrysocome* (J.R. Forster, 1781), since 1940 on Campbell Island (Cunningham & Moors 1994), Amsterdam Island (Guinard *et al.* 1998) and the Falklands (Bingham 1998), and associated declining oceanic productivity (Hilton *et al.* 2006), may relate in part to the position of the SAF.

Accidental introduction of Black Rats, *Rattus rattus* (Linnaeus, 1758), to Taukihepa (Big South Cape) in 1963 (Bell 1978) may have suppressed Sooty Shearwater populations (Scofield & Christie 2002, Moller *et al.* 2003b). Additionally, the introduction of Weka, *Gallirallus australis* (Sparrman, 1786), an endemic rail, to several Tītī Islands in the early twentieth century may have added to ongoing harvest pressure on chicks (see Kitson & Moller 2008). However, the factors regulating shearwater populations and potential compensation for these added predation and harvest impacts are unknown (Moller 2006) and, in general, the impacts of added chick losses are likely to be much lower than predation of adults (Hamilton & Moller 1995, Hunter & Caswell 2005, Wilcox & Donlan 2007, Finkelstein *et al.* 2008). If a high level of inter-island movement is typical for Sooty Shearwater populations, as seen for Short-tailed Shearwater (Skira 1991), The Snares population may be affected by population declines occurring elsewhere, potentially driven by predation by rats and Weka and harvesting of chicks.

New Zealand Sea Lions, *Phocarctos hookeri* (Gray, 1844), haul out up to 500 m from where they come ashore on The Snares, but reports of this species breeding here are rare. Sea Lions can compact the soil, crush burrow entrances and create wallows and mud pools. Most of our sampling units were situated within 500 m of Sea Lion haul-out locations. Populations of New Zealand Sea Lions are increasing (McNally 2001, 2002), and while we have little direct evidence of Sooty Shearwater failure attributable to sea lions, potential impacts on breeding success cannot be ruled out.

### Implications of Sooty Shearwater decline on the ecology of The Snares

Sooty Shearwaters are a keystone species in the ecology of The Snares and other islands on which they breed (Warham 1996, Moller *et al.* 2000, Hawke & Newman 2005). A decrease in burrow numbers will reduce soil aeration, erosion, evaporation of surface moisture, nutrient cycling, and composting, and may alter vegetation regeneration patterns. The impacts of a Sooty Shearwater decline on three threatened or endangered plants on The Snares (*Lepidium oleraceum* G. Forst., *Anisotome acutifolia* (Kirk) Cockayne and *Stilbocarpa robusta* (Kirk) Cockayne) are unknown. There are about 310 Southern Skua, *Catharacta lonnbergi* Matthews, 1912, on The Snares that consume a considerable number of Sooty Shearwaters (Miskelly *et al.* 2001). A reduction in overall Sooty Shearwaters may result in an increase in predation by skua on less common species. Reduced competition for burrows sites may allow marginalised species such as Mottled Petrels, Common Diving-Petrels, Broad-billed Prions, *Pachyptila vittata* (Forster, 1777), and Fairy Prions, *Pachyptila turtur* (Kuhl, 1820), to expand, and reduce egg failure in Snares Island Snipe *Coenocorypha aucklandica huegeli* (G.R. Gray, 1845) caused by Sooty Shearwater disturbance (Miskelly 1999).

## CONCLUSION

### Conservation status of Sooty Shearwater

The decline of Sooty Shearwater on The Snares is substantial and will potentially have far-reaching ecological consequences. One of the IUCN (2001) criteria used to class a species as "vulnerable" is a documented decline of 50% over the last 10 years or three generations (criterion A1, IUCN 2001). Our study documented the decline of Sooty Shearwater on North East Island during the past 27 years, between the initial and recent survey, but could not apportion the decline between the last 10 years and those preceding. Additionally, the 27-year period is equivalent to only one or two generations of this species. Therefore the documented decline does not meet the threshold specified under this criterion. An alternative criterion is a reduction of >30%, projected or expected within the next 10 years or three generations, whichever is the longer (criterion A3, IUCN 2001). The average rate of decline observed over the study interval exceeds the rate that would be necessary to halve the population in three generations, at least when our maximal estimate of generation time is used (21.6 years). The environmental precautionary principle suggests that we use this longer estimate of generation time for risk assessment. Even if the shorter estimate is used (14.7 years), the mean estimate is for a 53% decline over three generations, if the observed rate continues. These estimates clearly meet the requirements of this alternative criterion.

A more reliable long-term record of occupancy estimates will be crucial to determining whether decreases in burrow numbers translate to decreases in Sooty Shearwater breeding populations. Rigorous estimates of other Sooty Shearwater populations would also be beneficial for establishing whether the declines we observed on The Snares are occurring at the same rate throughout the species range, including the large breeding populations in Chile (Reyes-Arriagada *et al.* 2006). Recent reductions in shearwater bycatch (Uhlmann *et al.* 2005) and eradication of rats from four Sooty Shearwater breeding colonies (Moller *et al.* 2003b, Coote & Blackwell 2006) may slow past declines, but other threats like climate change and harvesting continue. Clearly there is cause for concern for this abundant and important island ecosystem engineer species and the conservation values it affects. Ongoing and intensified monitoring of species abundance throughout the range of the Sooty Shearwater is required.

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# NUTRIENT RESERVE DIFFERENCE BETWEEN YOUNG AND ADULT SHORT-TAILED SHEARWATERS, *PUFFINUS TENUIROSTRIS*, BEFORE AND AFTER TRANS-EQUATORIAL MIGRATION

by Nariko Oka

(with five text-figures and one table)

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The Short-tailed Shearwater, *Puffinus tenuirostris*, is a trans-equatorial migrant, breeding in Tasmania and other parts of southeastern Australia during the Austral summer and migrating to the northern North Pacific for the Boreal summer. Juveniles, in particular, suffer considerable mortality during the migration. Body composition of juveniles and adults in terms of body mass, lipid content, pectoral muscle mass and bone marrow content, before and after migration was investigated. Juveniles varied greatly in body composition pre- and post-migration and took longer to recover their nutritional status than adults. Adults sustained a better body composition, showing a moderate decrease in body fat upon arrival in the north and recovering their body mass and lipid supply during their stay. The lipid content of beachcast bird carcasses (mean, 4 g) was the minimum necessary for cell membranes and was not metabolisable for energy. Pectoral muscle protein remained high among birds under hyper-nutritional conditions, and decreased gradually at first as lipids decreased, and then rapidly at late malnutritional stages when most lipids had been utilised for energy. Although adults sustained their nutritional status, the amount of lipids in adults leaving the Tasmanian colony was insufficient to accomplish the long-distance migration, so post-breeding adults probably first visited Antarctic waters to accumulate fat reserves before commencing their northward migration.

**Key Words: age group, body composition, fat deposit, marrow, nutritive difference, pectoral muscle, protein, *Puffinus tenuirostris*, Short-tailed Shearwater, trans-equatorial migrant.**

## INTRODUCTION

Long-distance migrations through areas with poor nutrient productivity are critical events for some animals. Recent evidence has shown that the Sooty Shearwater, *Puffinus griseus* (J.F. Gmelin, 1789), a New Zealand trans-equatorial migratory seabird, traversed low-latitude waters with low productivity at mean speeds as high as 910 km per day while rarely foraging (Shaffer *et al.* 2006). To accomplish these long journeys without refueling, they must have sufficient energy reserves, though the storage abilities of most birds are limited by the constraints of flight and wing loading.

The Short-tailed Shearwater, *Puffinus tenuirostris* (Temminck, 1835), which breeds in Tasmania and other parts of southeastern Australia during the Austral summer (October to April), migrates one of the longest distances of any migratory bird, annually completing a trans-equatorial migration to spend the Boreal summer (May to September) in the highly productive waters of the northern North Pacific, Okhotsk Sea and Bering Sea (Serventy *et al.* 1971, Shuntov 1974, Warham 1996). This species has evolved a feeding apparatus that allows it to take small marine animals effectively (Morgan & Ritz 1982), which are sparsely distributed in tropical and sub-tropical oceans along its migratory pathway (Reid 1962). Thus, the survival of the migrants may depend greatly on energy deposits acquired before departure from the cool waters of the higher latitudes of both hemispheres.

The Nemuro Strait (about 44°N latitude), which separates eastern Hokkaido from the Kuril Islands, is the southernmost limit of the waters where the Short-tailed Shearwater spends the Austral winter. Along the Japanese coast, bathed by the warm Kuroshio Current 1300 km south of the strait, comprising the final stretch of the shearwaters' northward trans-equatorial migration, mortalities occur regularly during the two weeks from late May to early June (Oka *et al.* 1986).

Mass mortalities occurred at about 10-year intervals in the second half of the twentieth century: 1964, 1973–75, 1983–85 and 1996 (Kawaguchi & Marumo 1964, Sugimori *et al.* 1976, Oka & Maruyama 1986, Taguchi & Kato 1997). Age estimates based on skull ossification of the 2250 beachcast bird carcasses collected along the central Japanese coast from the mass mortality events of the mid-1980s revealed that almost all of them (99.5%) were four-month-old juveniles that had died from starvation (Oka & Maruyama 1986).

In the current century mass mortalities continued occurring along the central Japanese coast in the same period of late May–early June in 2003, 2005 and 2006, suggesting that those casualties were also migrating juveniles.

During the Boreal summer, mass mortalities of Short-tailed Shearwaters occurred in the Bering Sea in 1997 (Baduini *et al.* 2001a, b) and in the Okhotsk Sea in 2007 (Hokkaido Government, unpublished data). These mortalities suggest that some birds postponed recovering from energy shortage. Unfortunately the age of the birds in both these mass mortality events was not determined, but evidence from previous events suggests that it is more likely that they were inexperienced fledglings rather than experienced sub-adult or adult birds. As shearwater fledglings migrate independently to the Northern Hemisphere and remain apart from older birds while there, they have little foraging experience or knowledge of local waters suitable for feeding; thus, their survival is most likely lower than that of older birds, with differing energy storage among age groups.

Knowledge of energy storage differences among the shearwater age classes before and after the northern migration and the time to recover from energy shortage is important not only to clarify the reasons for the recent mortality

events observed in the Northern Hemisphere but also to understand the higher survivorship of adults (Bradley *et al.* 1989) from the viewpoint of nutritional ecology. I here report body composition differences in juvenile and adult Short-tailed Shearwaters and composition changes pre- and post-migration, focusing in particular on the disparity in nutritional recovery in the far northern waters of the Northern Hemisphere.

## MATERIALS AND METHODS

### Study animals

The Short-tailed Shearwater, *Puffinus tenuirostris*, is a long-distance migrant, which breeds in Tasmania and in other parts of southeastern Australia and overwinters in the North Pacific and Bering Sea and Okhotsk Sea (Marchant & Higgins 1990). The breeding season of Short-tailed Shearwaters comprises the 195 days from October to April (53% of each year; Warham 1990). The female lays a single egg in a burrow in late November (Marchant & Higgins 1990, Warham 1990). After the chick hatches in mid-January, the adults rear the nestling for about 85 days, until early April (Oka 1989). During this time, the chick increases its body mass until it weighs on average 1.4 times at maximum the weight of adults (Marchant & Higgins 1990). Beginning in early April, after provisioning the nestlings with a final meal, the adults leave their breeding grounds for the trans-Pacific migration, probably via Antarctic waters as a satellite tracking experiment suggests (Nicholls *et al.* 1998). During the subsequent two or three weeks, the body mass of the nestlings decreases, and they fledge in late April (Oka 1989), after their feathers have developed to adult condition (Lill & Baldwin 1983, Oka 1989). Most juveniles depart their hatching sites shortly after fledging (Naarding 1980). Since the juveniles depart later than the adults, they also arrive later at their overwintering sites in the North Pacific than the adults (Marchant & Higgins 1990). Large flocks of birds (presumably adults, judging from the departure schedule from the breeding grounds) first appear along northern Japanese coasts in the westernmost North Pacific in late April to early May (Niizuma & Shibano 1983, Watabe *et al.* 1987), and they first appear in the Bering Sea around the end of April (Shuntov 1974). Large flocks of fledglings appeared in Japanese waters from around 20 May to early June (Oka 1986). Adults leave the northern North Pacific and Bering Sea for their Australian breeding sites in September. They are followed by sub-adults and young one or two months later. Short-tailed Shearwaters are long-living seabirds, with 87–95% annual survivorship (Bradley *et al.* 1989). They begin to breed at about 6–7 years of age (Bradley *et al.* 1989).

### Materials

A total of 263 Short-tailed Shearwaters were used in this study: 31 birds (17 nestlings, 14 adults) were captured in burrows at the Clifton Bluff breeding colony in Tasmania, Australia (42°55'S, 147°19'E)(fig.1, location 1), in April 1983, weighed using a spring balance with 10 g calibration, and immediately released after these measurements were taken. A further 232 birds were collected from 4 April 1983–30 May 1984, euthanased (if not already dead) and frozen for later laboratory analysis. Of these birds 59 (50 adults, nine nestlings) were caught from burrows at their breeding grounds

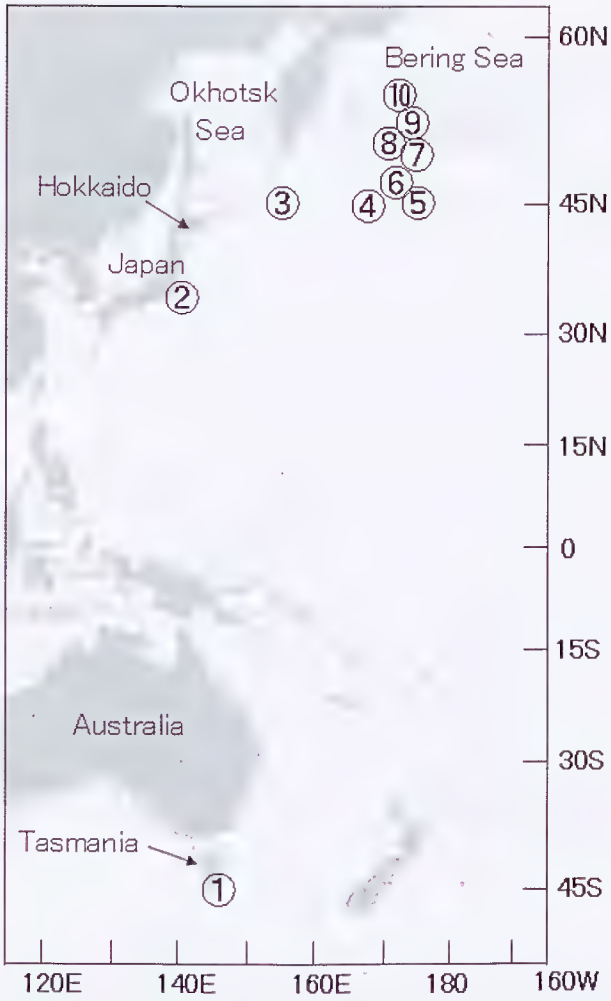


*FIG. 1 — Short-tailed Shearwater sampling locations.*

at the Clifton Bluff colony in Tasmania from 23 November 1983 to 20 April 1984; 60 fresh beachcast bird carcasses and 23 accidental bycatch birds from fishing nets were collected from Japan (collection points about 36°N in the North Pacific along the Japanese coast) (fig.1, location 2), from 24 May to 7 June 1983 and on 30 May 1984, when the shearwaters are at the final stretch of their migration; and 90 bycatch birds were collected from drift nets in the northern North Pacific and Bering Sea (between 44.50–55°25'N) from 5 May to 23 July 1983, where they spend the boreal summer (fig. 1, locations 3–10). Except for beachcast birds, all birds collected from the North Pacific waters and Bering Sea were bycatch, so they were alive at the time of capture.

### Age and phase groups

Birds were classified into two age groups: young (n = 147 with 26 nestlings and 121 juveniles) and adults (n = 116). Young birds were further classified into four phase groups: old nestlings (n =26) (captured in burrows in Tasmania in fledging period); dead juveniles (n = 60) from Japanese coasts (beachcast birds from around 36°N); juveniles (n = 23) collected in late May or early June upon their arrival (AR) at 45–48°N in the North Pacific, in the convergence zone of the warm Kuroshio Current with the sub-Arctic Oyashio Current; and juveniles (n = 15) collected from the

sub-Arctic Current waters north of 48°N to 55°N in the North Pacific and Bering Sea in July, during their stay (ST) in the Northern Hemisphere.

Adults collected from the North Pacific and Bering Sea during the Boreal summer were divided into two phase groups: those collected upon arrival (AR) in early to mid-May from the waters at 45–48°N in the North Pacific (n = 37), and those collected from sub-Arctic current waters north of 48°N to 55°N in the North Pacific and Bering Sea in late June to July during their Northern Hemisphere stay (ST) (n = 15). The timing of the AR and ST periods differed between the age groups, because, conservatively estimated, adults in Australia start their northward migration two weeks before the juveniles.

The ages of the birds collected outside the burrows at the colony were determined via skull ossification (Sugimori *et al.* 1986), which distinguishes juvenile Short-tailed Shearwaters from older birds. This method cannot be used to divide older birds into adults and sub-adults, so the adults collected from waters of the North Pacific may have included sub-adults.

## Laboratory procedure

The 232 birds used for the laboratory analysis were deep-frozen in zippered plastic bags soon after collection to prevent desiccation and were preserved in good condition until examination in 1984. Wet body and pectoral muscle masses and liver and stomach contents were weighed using a top-loading precision electronic balance (±0.01 g). Wet pectoral muscle weights were doubled for the analysis (i.e., full breast).

The carcasses were plucked by hand before dissection to prepare them for body lipid extraction. The following parts were removed from the naked body: the head (for aging); the bones of the legs and wings (for marrow analyses); the stomach and its contents; the intestines; the liver; and one side of the pectoral muscle. The liver and the one side of the pectoral muscles were first weighed and then individually minced by scissors and homogenised well; about 7 g of each sample was then collected for determination of the dry weight/wet weight ratio and the lipid and protein contents. The body carcass was minced, by passing it through a meat grinder three times using two kinds of grid sizes of blade, with hand mixing between passes to homogenise the sample. After thorough homogenisation, 7 g of each wet minced sample was individually weighed in the extraction thimbles using a top-loading precision electronic balance (±0.001 g) and dried in a convection oven at 50 °C for 8 hours until a constant weight was reached. The femora were cracked, and the marrow was removed as completely as possible from the bone with forceps and weighed. The marrow was dried to constant weight in a convection oven at about 50 °C for 3 hours, and the dry weight % (ratio of the dry weight to the wet weight) was recorded. Because not all measures were available for all birds sampled, sample sizes in the laboratory analyses differed from the numbers of birds collected.

Lipids were extracted from each dried sample by a Soxhlet extractor for 12 hours using an alcohol-benzene (1:1) solution. The lipid content and the lipid-free contents dried to a constant weight were weighed. The body lipid content given in this study is the total of the lipids extracted from the minced body and those in the liver and pectoral muscle side previously removed.

## Statistical analysis

Because no significant differences were found in any variable of the body composition of the 60 beachcast juvenile Short-tailed Shearwaters between 1983 and 1984 (table 1), I pooled the data for those two years. To test for significant differences in the beachcast juveniles between the two years, I used the Student *t*-test, as all variables were homoscedastic (body mass, $F_{10,48} = 0.530$ , $p = 0.2102$; body lipids, $F_{4,16} = 4.433$, $p = 0.0932$; dry body weight %, $F_{4,16} = 5.76$, $p = 0.5072$; wet pectoral muscle mass, $F_{10,48} = 0.4353$, $p = 0.5915$; protein content in pectoral muscles, $F_{5,23} = 0.838$, $p = 0.8070$; liver mass, $F_{10,33} = 2.024$, $p = 0.1810$; dry femur marrow weight %, $F_{10,47} = 1.018$, $p = 0.9714$; dry tibia marrow weight %, $F_{10,44} = 0.633$, $p = 0.3670$).

There were no significant differences in body mass in old nestlings (n = 26) in late April or in adults (n = 27) at the final landing period in late March and/or early April between 1983 and 1984 (between nestlings for two years: unpaired *t*-test, $t_{24} = 0.365$, $p = 0.7185$; between adults for two years: unpaired *t*-test, $t_{25} = 0.658$, $p = 0.5164$), using the Student *t*-test for significant differences, as variances

**Table 1**
**Comparison of body composition of beachcast birds in Japan between 1983 and 1984**

| | 1983 | | | 1984 | | | |
|---|---|---|---|---|---|---|---|
| | *n* | Mean | SD | *n* | Mean | SD | *p* |
| Body mass (g) | 11 | 297.3 | 17.54 | 49 | 285 | 24.09 | ns |
| Body lipids (g) | 5 | 4.79 | 2.881 | 17 | 4.29 | 1.368 | ns |
| Body lipids (%) | 5 | 1.63 | 0.905 | 17 | 1.45 | 0.367 | ns |
| Dry body weight (%) | 5 | 12.1 | 1.65 | 17 | 12 | 2.18 | ns |
| Pectoral muscle mass (g) | 11 | 26.6 | 4.13 | 49 | 27.5 | 5.03 | ns |
| Dry pectoral muscle content (%) | 11 | 22.7 | 1.99 | 47 | 22.5 | 1.73 | ns |
| Dry tibia marrow content (%) | 11 | 11.1 | 4.41 | 48 | 8.9 | 4.37 | ns |
| Dry femur marrow content % | 11 | 19.8 | 4.77 | 45 | 12.6 | 5.99 | ns |
| Liver weight (g) | 11 | 4.8 | 1.37 | 34 | 4.2 | 0.96 | ns |
| Pectoral muscle protein (g) | 6 | 4.4 | 1.08 | 24 | 5 | 1.18 | ns |
| Pectoral muscles/body mass (%) | 11 | 8.9 | 1.09 | 49 | 9.6 | 1.55 | ns |

were homoscedastic, respectively. I, therefore, used the data set of those birds collected in 1984 to compare the body composition variables among the different phase groups collected in 1983.

Because the body mass and body composition variables of both adults and young among three phase groups, before migration (i.e., shortly before leaving the colony in Tasmania), after migration (AR), and during their stay in the northern North Pacific and Bering Sea (ST), were heteroscedastic (i.e., with statistically unequal variances), I used the non-parametric Kruskal-Wallis rank test to determine significant differences in body mass and in body composition among these phase groups for both age groups. This was followed by non-parametric Mann-Whitney *U*-tests to compare the heteroscedastic variables between the two age groups. Statistical analyses were performed with Statview 5.0. All measurements in the text are given as means ± 1 standard deviation.

## RESULTS

### Body mass before and after migration

The mean body mass of young and adult Short-tailed Shearwater groups differed among the three phase groups, before migration (i.e., shortly before leaving the colony in Tasmania), after migration (AR) (i.e., shortly after arriving at the boreal summering waters), and during their stay in the northern North Pacific and Bering Sea (ST) (Kruskal-Wallis rank test, adults: $H = 22.37$, $p < 0.0001$, $n = 66$; juveniles: $H = 36.15$, $p < 0.0001$, $n = 55$) (fig. 2). The lowest mean body masses were found in both age groups upon arrival in the North Pacific (AR group). The body mass of adults was low (517 ± 31.4 g, $n = 37$) and that of juveniles was the lightest (397 ± 41.0g, $n = 26$) among all age and phase groups. During their stay in the northern North Pacific and Bering Sea in July (ST group), adults recovered their mean body mass to 583 ± 68.3 g ($n = 15$), a value significantly similar to that of adults at the final landfall in early April (AR group; Mann-Whitney *U*-test, $U = 91$, $z = -0.611$, $p = 0.5407$), whereas juveniles in the ST group showed only a moderate increase in body mass to 449 ± 61.5 g ($n = 12$) (Mann-Whitney *U*-test, $U = 75$, $z = -2.544$, $p = 0.011$), compared with the AR group, a value about 130 g lighter than the mass of adults spending the boreal summer in the northern North Pacific or Bering Sea and 160 g lighter than the mean value for old nestlings in fledgling period in Tasmania in late April.

### Relationship between body mass and body lipid content by weight

Body mass and the amount body lipids (g) were highly correlated ($r^2 = 0.79$, $p < 0.001$, $n = 153$) in the pooled data set of Short-tailed Shearwater age and phase groups ranging from 220–701 g in body mass, and from 2.6–149.3 g in body lipids, excluding nestlings. That was because the nestlings were in hyper-nutritive condition, characterised by a large amount of body lipids (262.3 ± 38.5 g, $n = 9$) accounting for 42.0% of their average body mass (622 ± 68.4 g, $n = 9$). The equation of the exponential correlation function is:

$$y = 10.932 - 0.151x + (4.564)^{-4}x^2$$

where $y$ is body lipids and $x$ is body mass. Body lipids reached a minimum of a few grams when the body mass had decreased to around 300 g or less (fig. 3).

### Relationship between body lipids and muscle protein

Body lipids and protein in the pectoral muscles were also highly correlated in the Short-tailed Shearwaters ($y = 4.65 + 0.285x - 0.002x^2$; $r^2 = 0.76$, $p < 0.001$, $n = 76$) where $y$ is protein in pectoral muscles (g) and $x$ is body lipids (g) (fig. 4).

Protein content was approximately constant at a high level in birds storing large amounts of fat, but decreased gradually as body lipids decreased below about 60 g. In birds that had lost most of their body lipids, the protein content had decreased to just a few grams (fig. 4).

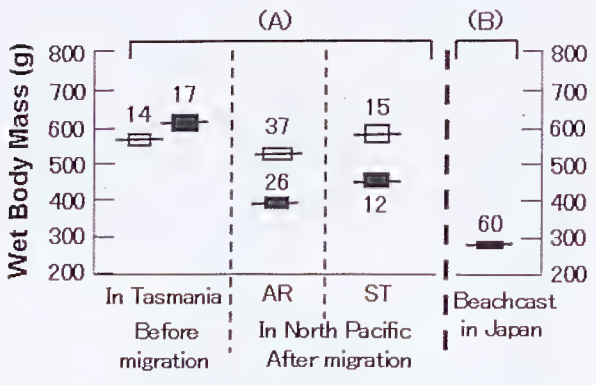


*FIG. 2 — Comparison of the body masses of young (black symbols) and adult (open symbols) Short-tailed Shearwaters before and after migration (A) and of beachcast birds that died during migration (B). AR indicates data collected at the time of the birds' arrival at the North Pacific for the Boreal summer, and ST indicates data collected later during their stay in the North Pacific. The mean (horizontal line) and standard error are shown.*

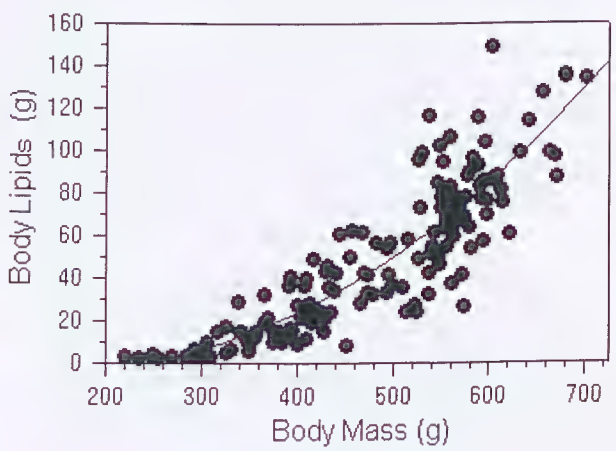


*FIG. 3 — Relationship between body mass and body lipid content in Short-tailed Shearwaters.*

### Beachcast birds

The mean body mass of beachcast young Short-tailed Shearwaters dying during migration was significantly lower than that of AR group (Mann-Whitney $U$ test, $U = 8.5$, $z = 7.255$, $p < 0.0001$) (fig. 2B). The beachcast birds retained only a mean lipid content of 4.4 ± 1.74 g (range, 2.62–9.84 g, $n = 22$) (fig. 5), accounting for a mean 1.5 ± 0.51% (range, 1.1–3.2%, $n = 22$) of the very lean average body mass of 287 g. They also had a small pectoral mass (fig. 5B); the pectoral muscle weighed 27.3 ± 4.86 g (range, 17.2–41.1 g, $n = 60$), with only 4.9 ± 1.17 g of protein (range, 3.4–8.2 g, $n = 30$). Liver mass was 4.6 ± 1.25 g (range, 3.0–8.3 g, $n = 45$) (fig. 5B). The marrow content of the dried femur and tibia was only 14.0 ± 6.43% (range, 5.0–32.7%, $n = 56$) (fig. 5B) and 9.8 ± 4.85% (range, 1.9–26.4%, $n = 59$), respectively. These low marrow values indicated that the birds were alive until the suspension of their hematopoietic stem cell activities.

All of the lines of evidence showed that the beachcast young Short-tailed Shearwaters had exhausted their nutrients to critical levels by the time they had reached the Japanese coast, which is the final and westernmost stretch of their migration pathway to the northern North Pacific.

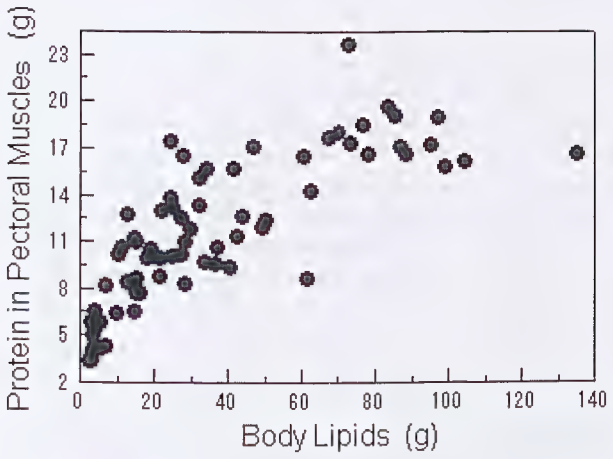


*FIG. 4 — Relationship between the protein content in pectoral muscles and the body lipid content in Short-tailed Shearwaters.*

## BODY COMPOSITION

### Post-migratory birds

Post-migratory AR young stored 26.2 g (± 15.22 g, $n = 23$) of body lipids (fig. 5), which accounted for 6.6% of the lean average body mass of 397 g, whereas post-migratory AR adults stored 41.0 g (± 13.03g, $n = 13$) of body lipids, accounting for 7.9% of the average 517 g of body mass. The amount of fat stored by the post-migratory young was therefore only 10% of that of the obese nestlings sampled during the fledging period at the colony in Tasmania (mean body lipids, 262.3 g, 42% of the 622 g body mass; 20 April 1984) (fig. 5). Moreover, upon their arrival in the northern North Pacific (AR group), juveniles had similar fat levels to juveniles that were collected later, i.e., during their stay in the North Pacific and Bering Sea (unpaired $t$-test, $t_{30} = 0.066$, $p = 0.948$). On the other hand, post-migratory adults stored 71.4 g (±30.94g, $n = 15$) lipid, that was similar to the body lipids stored by pre-migratory adults leaving the colony in Tasmania in 1984 (84.4 ± 16.15 g, $n = 13$)(Mann-Whitney $U$ test, $U = 71$, $z = -1.221$, $p$=0.222). Though the pectoral muscle mass decreased significantly even in the adults from before to after migration (unpaired $t$-test, $t_{48} = 2.703$, $p = 0.01$), the decrease was moderate (<10%) (fig. 5).

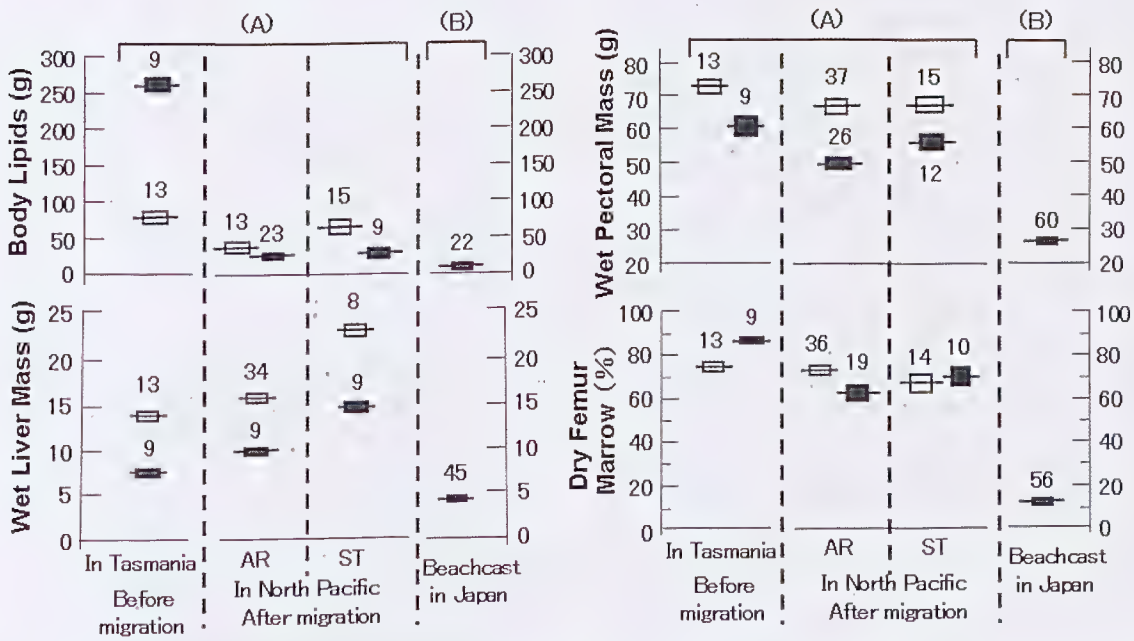


*FIG. 5 — Comparison of the body masses of young (black symbols) and adult (open symbols) Short-tailed Shearwaters before and after migration (A) and of beachcast birds that died during migration (B). AR indicates data collected at the time of the birds' arrival at the North Pacific for the boreal summer, and ST indicates data collected during their stay in the North Pacific. The mean (horizontal line) and standard error are shown.*

The fat levels after migration (AR groups) were larger in both young (26.2 ± 15.21 g, *n*=23) and adults (41.0 ± 13.03 g, n=13) than the mean minimum value found in the dead birds (beachcast birds: 4.4 g, n=22). Their residual lipid capacities therefore were sufficient for use not only for energy for basal metabolism but also for foraging in local waters. Although the AR young had lost their nutritive reserves to some degree during migration, they sustained high contents of dry femur marrow (AR group, 62 ± 21.5%, $n$ = 19; ST group, 70 ± 17.5%, $n$ = 10) (fig. 5) that did not differ significantly from the adult AR (mean, 72.9 ± 6.23%, $n$ = 13) and ST (mean, 68.0 ± 19.16%, $n$ = 14) groups (Kruskal-Wallis rank test, $K_5$ = 3.684, $p$ = 0.595).

Adults in the northern North Pacific in late June to July (ST group) had the largest wet liver mass (23.0 ± 2.55 g, $n$ = 8) among all phase and age groups (fig. 5), indicating a high intake efficiency while foraging.

## DISCUSSION

Body lipids in Short-tailed Shearwaters are both a structural component necessary for cell membranes, and thus not available as a metabolic energy source, and a stored body fat component, including a distinctive oil stored in the upper layer of the stomach, which can be mobilised as a metabolic energy source.

The data presented here on lipid content from beachcast juvenile birds and from adults at the breeding colony led to a number of important observations.

The juvenile Short-tailed Shearwaters beachcast along Japanese shores during the northern migration had a remaining mean body lipid content of 4 g, which is comparable to the mean value of 3 g in starved birds of unknown age in the Bering Sea, reported by Baduini *et al.* (2001b). Oka & Okuyama (2000) reported that beachcast Rhinoceros Auklets, *Cerorhinca monocerata* (Pallas, 1811), found after an oil spill, had a mean body lipid content of 7 g and dry femur marrow content of 24%. These auklets, which are short-distance migrants and a slightly heavier species than the shearwaters, had also lost most of their nutritional energy reserves and had suspended hematopoietic stem cell activity.

The body lipids remaining in the beachcast Short-tailed Shearwaters, therefore, should not be regarded as available for metabolic energy needs but as structural components of cell membranes (Klasing 1998). Thus, this species of bird requires extra lipids for energy, in addition to the 4 g needed for structural components. The beachcast juveniles also had only a small amount of protein in their muscles, as has been found in starved penguins (Cherel *et al.* 1994). The present study also shows that protein had a tendency to be consumed at the malnutritional stage after body lipid was mostly consumed.

The two mean body mass values (564 g with empty stomachs, and 569 g, possibly including stomach contents) for the adult birds collected in this study at the colony in southern Tasmania during the departure period compares well with adult values measured by other studies at colonies during the late chick-rearing season (e.g., 574 g mean body mass with empty stomachs; northernmost Montague Is. population, New South Wales, Schultz & Klomp 2000; and 583 g mean body mass, possibly including stomach content; Cape Woolamai population on Phillip Island, Victoria, Lill & Baldwin 1983). The data can thus be interpreted as indicating the standard nutritive condition of adults.

As body mass and body fat have been previously shown to be positively correlated in Rhinoceros Auklets (Oka & Okuyama 2000) and Short-tailed Shearwaters (Baduini *et al.* 2001b), consistent with the results of the present study, the moderate body mass of 564 g with a mean body fat content of 83 g can be considered optimal from the viewpoint of keeping wing loading at a minimum to increase flight efficiency. This stable nutritive condition among adults is maintained by commuting from the breeding sites far south to abundantly productive Antarctic waters, where Antarctic Krill, *Euphausia superba* (Dana, 1852), is captured to replenish their energy reserves during the chick-rearing season (Weimerskirch & Cherel 1998, Klomp & Schultz 2000, Cherel *et al.* 2005). Even if the birds minimise energy expenditure during flight by dynamic soaring, their fat deposits are too small to accomplish the 9000-km long-distance migration from 42°S to 45°N through the low-productivity low-latitude waters. During the final long-distance foraging trip to Antarctic waters south of Tasmania prior to the equatorial migration, adults may replenish their fat deposit and distinctive stomach oil to some extent. During the trans-equatorial migration over low-productivity tropical waters, the birds would metabolise not only their stored fat but also some of the stomach oil for their energy requirements. Birds probably complete their trans-equatorial migration in minimal time to avoid wasting their limited amount of energy on opportunistic foraging efforts in areas of low productivity.

Body lipids reached a minimum of a few grams when the body mass had decreased to around 300 g or less. This suggests that the decrease in body mass during the final stages of malnutrition may reflect decreases in components other than body fat. Unlike lipid, protein generally is not viewed as an energy storage, but is considered to be used only after lipid reserves are exhausted (Blem 1990). Protein content in the pectoral muscles in the shearwaters was accelerated to decrease during the final critical stage of malnutrition. Though the pectoral muscle mass decreased significantly even in adults from before to after migration, the decrease was moderate, compared with the decrease in body lipid, that showed Short-tailed Shearwaters depend largely on body lipids to accomplish their long-distance migration.

Post-migratory juveniles sustained a low body mass with a small amount of body lipids. The results of this study indicate that juveniles take longer to recover from energy shortages than adults, possibly because they lack skill and experience in feeding on prey organisms.

During anomalous weather and oceanic conditions in the Bering Sea in 1997 and 1998, the shearwaters failed to accumulate sufficient energy, storing only a mean of 31 g and 45 g body fat by each North Pacific autumn, respectively. They also had lighter pectoral masses (Baduini *et al.* 2001a) at the time when the shearwaters usually commence their southward migration to Australia. In the North Pacific autumn of 1997, up to 200 000 individuals were estimated to have starved to death (Baduini *et al.* 2001b). Interestingly, these fatalities had body and pectoral masses 25% larger than those of the beachcast juveniles along the Japanese coast in this study (380 g *vs.* 287 g for body mass, and 36 g *vs.* 27 g for pectoral mass, respectively). The ages of the dead birds in the 1997 mass mortality were not determined.

Comparison of the body and pectoral masses of these fatalities with those of the adult birds in this study suggests that many older age classes were involved in these events, as

the structural weights of bone and organs such as muscles increase and develop with age. These mass mortalities in the Bering Sea differ ecologically from the mass mortalities in temperate waters in which migratory juveniles are repeatedly involved. The ages of the dead birds should be determined to understand the effects of these mass mortalities on the population ecology of Short-tailed Shearwaters, and long-term monitoring is essential to understand the nature of these events.

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## Citation format

- published material e.g., Brown (1983), Brown & Green (1985), Brown *et al.* (1987), Brown *et al.* (in press), (Brown 1983)
- note any information supplied personally to the author(s) e.g., (T.O. White pers. comm.); do not include in the list of references; obtain permission from the informant (preferably in writing) to quote him or her
- include the relevant page number of a lengthy cited work e.g., (1983: 110)
- do not refer to papers "in preparation"; cite these as (author(s) unpubl. data).

## References format

The reference list should conform to the format used in recent volumes of the journal. Common examples are as follows:

Single author paper:

**Bingham, M.** 1998: The distribution, abundance and population trends of Gentoo, Rockhopper and King penguins in the Falkland Islands. *Oryx* **32**: 223–232.

Two author paper:

**Hawke, D.J. & Newman, J.** 2005: Using isotope analysis to identify incorporation of marine nutrients in terrestrial birds at Snares Islands. *Notornis* **52**: 108–111.

Multiple author paper:

**Ainley, D.G., Veit, R.L., Allen, S.G., Spear, L.B. & Pyle, P.** 1995: Variations in marine bird communities of the California current, 1986–1994. *California Cooperative Oceanic Fisheries Investigations Reports* **36**: 72–77.

Book:

**Davison, A.C. & Hinkley, D.V.** 1997: *Bootstrap Methods and their Application.* Cambridge University Press, Cambridge: 592 pp.

Edited work:

**Marchant, S. & Higgins, P.J.** (eds) 1993: *Handbook of Australian, New Zealand and Antarctic Birds. Volume 2: Raptors to Lapwings.* Oxford University Press, Melbourne, Australia: 1048 pp.

Chapter in an edited work:

**Artyukhin, Y.B. & Burkanov, V.N.** 2000: Incidental mortality of seabirds in the drift net salmon fishery by Japanese vessels in the Russian Exclusive Economic Zone, 1993–1997. *In* Kondratyev, A.Y., Litvineko, N.M. & Kaiser, G.W. (eds): *Seabirds of the Russian Far East.* Canadian Wildlife Service Special publication, Ottawa: 105–116.

Thesis:

**Charleton, K.J.** 2002: Variation in Sooty Shearwater burrow entrance density, burrow use and chick emergence: Science and Traditional Environmental Knowledge approaches. Unpublished MSc thesis, University of Otago, Dunedin.

Electronic version of journal:

**Chiswell, S.M. & Rickard, G.J.** 2006: Comparison of model and observational ocean circulation climatologies for the New Zealand region. *Journal of Geophysical Research* **111**: C10011, doi: 10.1029/2006JC003489.

Website:

**IUCN** 2001: *IUCN Red List of Threatened Species 2001 Categories & Criteria (version 3.1),* www.iucnredlist.org (accessed 22 January 2007).

Report:

**MacKenzie, D.I. & Fletcher, D.** 2005: Characterisation of seabird captures in NZ fisheries. Report to Ministry of Fisheries for Project ENV2004-04. April 2006. Unpublished Report to the Ministry of Fisheries, Wellington: 99 pp.

PLEASE NOTE that an en-dash should be placed between page numbers NOT a hyphen.

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- number consecutively in the order of the first reference to them in the text

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The preferred dimensions for tables are either (a) 65 lines or less in depth, each line of no more than 130 characters and spaces in length, or (b) 42 lines or less in depth, each line of no more than 190 characters and spaces. These dimensions include the title, explanatory material and column headings.

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Material presented as Appendices (labelled 1, 2, 3, etc.) is accepted at the editor's discretion, but may be printed in a smaller type than the body of the paper.

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The Botanical, Zoological and Stratigraphic Codes of Nomenclature should be fully observed.

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